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Evaluating solids, phosphorus, and nitrogen cycling and transport in vegetative treatment systems used for runoff control on beef feedlots

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Evaluating solids, phosphorus, and nitrogen cycling and transport in vegetative treatment systems used for runoff control on beef feedlots

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Abstract

Runoff from open lot animal feeding operations has been recognized as a potential pollutant to receiving surface waters. This effluent is known to contain nutrients such as nitrogen and phosphorus, as well as other potential pollutants such as organic matter, solids, and pathogens. Increased environmental awareness has prompted the need for improved feedlot runoff control. As a result, open feedlots of all sizes are looking for cost effective alternatives to handle feedlot runoff. Vegetative treatment systems (VTSs) have been proposed as a potential option to control this runoff, enhance environmental security, and protect water quality. Although previous research has shown that vegetative treatment systems can be effective in plot-scale and limited field-scale studies, questions about their performance on commercial operations remain. In addition the sustainability and the mechanisms by which treatment is occurring are still uncertain. Answering questions about the mechanisms these systems use to provide treatment offers the possibility of improvements in future designs and will increase our ability to effectively operate and manage existing systems. Thus, the objectives of this research was to evaluate solids, phosphorus, and nitrogen transport and cycling within the vegetative treatment system to better understand that fate of these contaminants, and in doing so to improve the design and management of vegetative treatment systems.

This dissertation will consist of work on each of these areas, solids, phosphorus, and nitrogen transport and cycling, as they, along with hydrology, are the keys to understanding vegetative treatment system performance and sustainability. The first section, on solids transport consists of three manuscripts. The first manuscript, "Using total solids concentration to estimate nutrient content of feedlot runoff effluent from solids settling basins, vegetative infiltration basins, and vegetative treatment areas," relates nutrient content in feedlot runoff from solid settling basins, vegetative infiltration basins, and vegetative treatment areas to the solids content within the effluent. This analysis serves the purpose of demonstrating that managing and understanding the sedimentological connections within the treatment system provides a great deal of insight into transport of other parameters (particularly nitrogen, phosphorus, and organic matter). Specifically, this work demonstrates that if detailed models of sediment export from the feedlot and through the treatment system can be developed, then this information can be used in predicting the movement of other parameters of concern. The second manuscript, "A review of settling characteristics of solids in runoff from beef feedlots" reviews the sediment characteristics that are required to perform detailed modeling of solids transport within the treatment system. Specifically, the manuscript reviews the physical characteristics (particle size, density, and settling rates) of particles transported in runoff

from beef feedlots, addressing how these properties differ between various feedlots with different surface conditions (concrete and earthen) and at different locations. The review focuses on the implications these settling properties have for designing successful sedimentation systems and in predicting the actual performance of settling basins. The third manuscript, “Development of a runoff and sediment routing model for open lot beef feeding facilities” describes the development of a hydraulic and sediment routing model designed to predict solids transport from feedlot surfaces. This model can be used for prioritizing feedlots that are in need of enhanced runoff control systems, evaluating the hydraulic and sediment loadings that a feedlot runoff control systems are required to handle, and for exploring how different feedlot sizes, layouts, and designs impact solids transport.

The second section, on phosphorous fate and cycling in the vegetative treatment areas, consists of a series of three manuscripts that utilize different monitoring procedures and assays to assess mechanisms of phosphorus treatment and its fate within the vegetative treatment area. The first manuscript uses a phosphorus mass balance approach to project phosphorus accumulation in the soil and compares the projected increases to monitored trends in soil test phosphorus at six vegetation areas in Iowa. The manuscript provides a preliminary phosphorus balance at six vegetative treatment areas focusing on how phosphorus is partitioning between soil, water, and vegetation. The second manuscript builds on this work by utilizing a sequential fractionation procedure, the Hedley method, to better understand the accumulation patterns of phosphorus within the soil and thereby obtain the relative stability of the accumulated phosphorus. Results of the fractionation procedure were interpreted based on the concept that a maximum soil phosphorus retention capacity existed; however, none of the soils as of yet exhibited a phosphorus accumulation pattern indicative of saturation, although in many cases, specific pools, mostly organic phosphorus pools, did appear saturated. The third manuscript utilizes a phosphorus sorption experiment to evaluate how the soil’s phosphorus retention properties had been modified by five years of use as vegetative treatment areas. Specifically, the experiment evaluated how continued use of the vegetative treatment area modified the soil properties and the impact this had on the estimated phosphorus sink capacity of the soil. This experiment provides an evaluation of whether the life expectancy model developed previously by Baker et al. (2010) provides a useful estimation of vegetative treatment area phosphorus saturation life and explores what mechanisms may be allowing further phosphorus accumulation.

Finally, the third section, on nitrogen transport and cycling, contains two manuscripts. The first manuscript, “Vegetative treatment system impacts on groundwater quality,” discusses groundwater concentrations up-gradient, within, and down-gradient of six vegetative treatment system on beef feedlots in Iowa. The manuscript provides statistical comparisons and trend analysis to evaluate impacts the system may be having. Nitrate leaching in the vegetative treatment system is also estimated. The second and final manuscript, “The impact of vegetative treatment area use on soil biologically available carbon and nitrogen pools,” reports the results of a long-term carbon and nitrogen fractionation procedure to evaluate if accumulation of labile carbon and nitrogen is occurring and if this organic matter is nitrogen enriched. A final conclusions manuscript, “Vegetative treatment systems: design, management, and siting recommendations” provides recommendations on what is required to construct successful vegetative treatment systems and which areas require future research so that designs can be refined and ensure appropriate nutrient cycling and retention.

Chapter 1. General Introduction

INTRODUCTION

Runoff from open lot animal feeding operations has been recognized as a potential pollutant to receiving surface waters. This effluent is known to contain nutrients such as nitrogen and phosphorus, as well as other potential pollutants such as organic matter, solids, and pathogens, and as such is of concern to water quality. Due to increased recognition of the potential impacts feedlots can have on water quality and growing environmental awareness, cattle producers are facing increasing pressures to improve their feedlot runoff control systems. As a result, open feedlots are looking for cost effective alternatives to handle feedlot runoff in effective and environmentally sustainable ways. Vegetative treatment systems (VTSs) have been proposed as a potential option (Woodbury et al., 2003).

Vegetative treatment systems are wastewater treatment systems that use at least one form of vegetative treatment, (i.e. a vegetative treatment areas (VTA) or a vegetative infiltration basins (VIB)), with other pretreatment components to control and treat feedlot runoff (Koelsh et al., 2006). A sloped VTA is an area level in one dimension (width), to encourage sheet flow, and has a slight slope (less than 5%) in the other dimension (length) that is planted and managed to maintain dense, permanent vegetation (Moody et al., 2006). Operation of the VTA involves applying effluent evenly across the top edge of the VTA. The effluent then gravity-flows down the length of the VTA, where it is treated via sedimentation and infiltration. A VIB is a relatively flat area surround by berms to prevent surface outflow of effluent and underlain by drainage tiles, approximately 1.2 m below the surface, to maximize the infiltration of effluent into the soil. A VIB is periodically inundated with effluent to evenly distribute effluent over its surface. In the VIB effluent is subject to treatment via filtration as it drains through the soil, sorption of contaminants to the soil particles, and microbial actions the break down organic matter. Effluent draining through the soil profile is collected in the tile lines and pumped onto a VTA for further treatment.

Current federal Environmental Protection Agency (EPA) regulations allow the local government, typically the state government, to write and enforce the Concentrated Animal Feeding Operation (CAFO) runoff control guidelines. This is done through the issuance of a National Pollutant Discharge Elimination System (NPDES) permits. Feedlots in Iowa that use VTSs were granted NPDES permits so the performance of these alternative treatment systems could be evaluated. As part of their interim NPDES permits, feedlots in Iowa were required to compare the monitored mass of contaminant released from their VTS to the modeled mass of a properly designed and managed contaminant a containment basin runoff control system would have released. In Iowa, site-specific containment system performance was predicted using the Iowa State University Effluent Limitations Guideline model (ISU-ELG model) implemented according to the guidelines described in Appendix A of the Iowa AFO/CAFO Regulations (Iowa DNR, 2007).

In addition to the challenges facing CAFO sized operations, smaller feedlots (< 1000 head) are facing increasing scrutiny of their environmental stewardship as well, specifically in control of feedlot runoff. Although, these operations are typically not permitted under the NPDES system, they are required to settle all solids and avoid discharge of effluent to waters of the state, either directly or through man-made conveyances. As a result, these operations are seeking information on appropriately sizing runoff control systems to minimize the risk of water quality violations resulting from feedlot runoff. The smaller size of these operations limits their ability to leverage the economies of scales that CAFOs utilize in justifying containment basins. This has lead researchers and policy makers to suggest that due to the lower construction costs, vegetative treatment systems may be a potential options to provide these smaller operations with cost-effective runoff control options (Bond et al., 2011).

In a literature review, Koelsch et al. (2006) reported on approximately 40 field and 58 plot studies demonstrating that VTSs may be effective in a variety of situations; however, none of these studies where performed at commercial feedlots, but where instead performed at government and university research facilities. In their review Koelsch et al. (2006) reported that VTSs commonly reduced total solids transport from the feedlot in overland flow by 70-90%. Similarly, total nitrogen (TN), total Kjeldahl nitrogen (TKN), and ammoniacal nitrogen (NH_3/NH_4-N) were reduced by approximately 70% in properly designed and managed (VTA:Feedlot area ratios >1, maintenance of sheet flow, pretreatment via solids settling, and maintaining a stand of dense vegetation) VTAs (Ikenberry and Mankin, 2000). Phosphorus (P) removal rates were much more variable, with typical removal rates ranging from 7 to 100% (Koelsch et al., 2006). Although this previous research has shown that VTSs can be effective in plot-scale and limited field-scale (limited to university and government research facilities) studies, questions about their performance on commercial operations remain. In addition, the treatment mechanisms and operational lifetimes of these systems are still uncertain. Elucidating the treatment mechanisms offer the possibility to improve future designs and will increase our ability to effectively operate and manage existing systems. Such knowledge is also necessary to provide a complete assessment of VTS economics, which will help clarify the role these systems will play in addressing water quality issues facing animal agriculture.

OBJECTIVES

The objectives of this research were to evaluate solids, phosphorus, and nitrogen transport and cycling within the VTS to better understand the fate of these contaminants, and in so doing, to improve the design and management of VTSs. This dissertation consists of work in each of these areas (solids, phosphorus, and nitrogen transport and cycling) as they, along with hydrology, are the keys to understanding vegetative treatment system performance and sustainability. Objectives for each section are discussed below and are brought together holistically in a conclusions chapter that provides updated guidance on designing and managing VTSs and on future research needs.

Solids Transport and Removal in Vegetative Treatment Systems

1. Develop a model of the feedlot surface that is capable of predicting runoff volumes and sediment transport from the feedlot that can be used to aid in the design of runoff control systems and in evaluating the potential risk feedlots pose to surface waters.
2. Evaluate the settling characteristics of solids in runoff from Iowa beef feedlots and summarize the measured settling characteristics in comparison to literature values with specific emphasis on improving current settling basin design recommendations.
3. Determine whether total solids concentrations can be used as a proxy for nutrient content and effluent quality of feedlot runoff from solid settling basins, vegetative infiltration basins, and vegetative treatment areas.

Phosphorus Accumulation and Saturation

1. Develop a phosphorus mass balance model that can be used to aid in sizing vegetative treatment areas and project phosphorus accumulation. Evaluate performance of the model based on measured phosphorus loadings and measurements of soil Melich-3 phosphorus concentrations in the vegetative treatment area soils. Provide a mass balance to suggest fate of phosphorus applied to the VTA.
2. Determine phosphorus sorption parameters from vegetative treatment area soil before and after five years of use to evaluate how continued use as an effluent disposal area has affected soil phosphorus buffering capacity, equilibrium phosphorus concentration, and the overall maximum amount of phosphorus the soil could accumulate.

3. Perform a phosphorus fractionation procedure to investigate how the use of the soil as a vegetative treatment system has impacted the distribution of phosphorus within the soil and the lability of the phosphorus.

Nitrogen Retention and Cycling in Vegetative Treatment Areas

1. Statistically evaluate the impact VTS installation and use had on groundwater quality, focusing on nitrate, ammoniacal nitrogen, chloride, and fecal coliform concentrations. Estimate nitrogen leaching based on measured groundwater contaminant concentrations and a hydraulic balance.
2. Evaluate the impact that use of the vegetative treatment area had on biologically available soil carbon and nitrogen content as compared to the paired grass areas at each of the VTS locations.
3. Perform an estimated nitrogen balance based on measurements of surface inflows and outflows from the vegetative treatment areas, the amount of nitrogen accumulated in the soil (based on objective two), the amount removed with harvested vegetation, and the amount estimated to be leached (based on objective 1). Nitrogen not accounted for in this estimate will be considered to be lost as a gaseous emission, either through ammonia volatilization, nitrous oxide generation during nitrification, or as nitrous oxide and nitrogen gas generated during denitrification.

THESIS ORGANIZATION

The research presented in this dissertation is comprised of eight manuscripts broken into three sections (solids, phosphorus and nitrogen) with three manuscripts in the solids and phosphorus sections, two manuscripts in the nitrogen section. Basic conclusions for each manuscript are provided within the chapter, while broader conclusions and implications on design, management, and siting of vegetative treatment systems are provided in the final chapter of the dissertation.

Introduction to Solids Transport from Feedlots and Retention in Vegetative Treatment Systems

Sediments and solids play a crucial role in water quality; they can serve as either a sink or source of nutrients and contaminants depending on the environmental conditions. Specifically, nutrient enriched soils, such as feedlot surfaces, can serve as a source of easily erodible solids that have the potential to reach waterways and degrade water quality. However, practices implemented downstream of the enriched soil (filter strips, infiltration areas, sediment ponds, terraces, etc.) can interrupt flow patterns, cause settling of solids, and as a result, disrupt or break the hydrologic and sediment connection between the nutrient enriched soil and downstream surface waters. This hydraulic/sediment disruption is particularly important at concentrated animal feeding operations where the production area contains byproducts associated with animal production (manure and feedstuffs) that are rich in nutrients and organic matter, and due to their exposure to the elements, available for transport. The following three chapters explore these concepts by focusing on erosion of sediment from the feedlot surface, the physical and chemical properties that control the settling characteristics of the contaminants, and relating the transport of sediment to other parameters of interest to water quality (nitrogen, phosphorus, oxygen demand, etc.).

The first manuscript, "Using total solids concentration to estimate nutrient content of feedlot runoff effluent from solids settling basins, vegetative infiltration basins, and vegetative treatment areas," which was published in *Applied Engineering in Agriculture*, relates nutrient content in feedlot runoff from solid settling basins, vegetative infiltration basins, and vegetative treatment areas to the solids content within the effluent. This analysis serves the purpose of demonstrating that managing and understanding the sedimentological connections within the treatment system provides a great deal of insight into transport of other parameters, particularly nitrogen, phosphorus, and organic matter. Specifically, this work demonstrates that if detailed models of sediment export from the feedlot and through the treatment system can be developed, then this information can be used in predicting the movement of other parameters of concern and to predict the risk feedlot runoff poses to water quality. The second manuscript, "A review of settling characteristics of solids in runoff from beef feedlots" which will be submitted to *Transactions of the ASABE*, reviews the sediment characteristics that are required to perform detailed modeling of solids transport within the treatment system. Specifically, the manuscript reviews the physical characteristics (particle size, density, and settling rates) of particles transported in runoff from beef feedlots, addressing how these properties differ between various feedlots with different surface conditions (concrete vs. earthen) and at different locations. The review focuses on the implications these settling properties have for designing successful sedimentation systems and in predicting the actual performance of settling basins. The third manuscript, "Development of a runoff and sediment routing model for open lot beef feeding facilities" which also will be submitted to *Transactions of the ASABE*, describes the development of a hydraulic and sediment routing model designed to predict solids transport from feedlot surfaces. This model can be used for prioritizing feedlots that are in need of enhanced runoff control systems, evaluating the hydraulic and sediment loadings that a feedlot runoff control systems are required to handle, and for exploring how different feedlot sizes and layouts impact solids transport.

Phosphorus Fate and Retention in Vegetative Treatment Areas for Feedlot Runoff Control

Phosphorus is an important nutrient input for crop and livestock production; however, excessive losses to surface water can accelerate eutrophication and degrade water quality. This duality as resource and pollutant complicates phosphorus management. Specifically, the direct cost of phosphorus lost from agricultural systems is relatively small compared to potential monetary benefits of increased production; however, even low levels of phosphorus loss can have large effects on water quality downstream. This problem has taken on increased importance due to continued increases in quality of life for much of the world, and with it increased demand for meat, which has placed increased production demands on arable land.

Arable areas are often spatially separated from the concentrated livestock production systems, which limits recycling of phosphorus present in agricultural waste products. Thus, the issue of phosphorus management is especially relevant in animal agriculture and around municipal wastewater treatment plants, where due to the relatively dilute nature of the waste products and the high costs associated with their transport, phosphorus application often is in excess of crop phosphorus removal. This can result in increased soil test phosphorus and eventually leaching of soluble P or erosion of phosphorus enriched soil particles. This has lead to interest on the sustainability, life expectancy, and effectiveness of different wastewater treatment systems in terms of phosphorus management. Evaluation of these parameters for vegetative treatment systems used for feedlot runoff control is an important step in evaluating the overall benefits of vegetative treatment systems and comparing them to other waste management alternatives.

This section (chapters 5-7) explores these concepts focusing on the fate of phosphorus in vegetative treatment systems used for feedlot runoff control and the sustainability of the treatment mechanisms utilized. It consists of a series of three manuscripts that utilize different monitoring procedures and assays to assess mechanisms of phosphorus treatment and its fate within the vegetative treatment area. The first manuscript, to be submitted to the *Applied Engineering in Agriculture*, uses a phosphorus mass balance approach to project phosphorus accumulation in the soil and compares the projected increases to monitored trends in soil test phosphorus at six vegetation areas in Iowa. The manuscript provides a preliminary phosphorus balance at six vegetative treatment areas focusing on how phosphorus is partitioning between soil, water, and vegetation. Changes in soil test phosphorus in shallow surface soil (0-30 cm) were evaluated in the context of the phosphorus mass balance and were utilized to evaluate phosphorus distribution over the vegetative treatment area. Deep soil sample

(0-122 cm) results are presented to evaluate phosphorus movement with depth in the soil profile over the course of a 4-year monitoring period.

The second manuscript, to be submitted to *Soil Science*, builds on the previous work by utilizing a sequential fractionation procedure, the Hedley method, to better understand the phosphorus accumulation patterns within the soil and the relative stability of the accumulated phosphorus. The method used allows determination of which pools are accumulating phosphorus, how the phosphorus is partitioning among the soil pools, provides information on the stability of the incorporated phosphorus, and offers insight into the sustainability of the soil treatment mechanism. Results of the fractionation procedure were interpreted based on the concept of a maximum soil phosphorus retention capacity; however, none of the soils as of yet exhibited a phosphorus accumulation pattern indicative of saturation, although specific pools, mostly organic phosphorus pools, did appear saturated. The third manuscript, to be submitted to the *Soil Science Society of America Journal*, utilized a phosphorus sorption experiment to evaluate how the soil's phosphorus retention properties had been modified by five years of use as vegetative treatment areas. Specifically, the experiment evaluated how continued use of the vegetative treatment area modified the soil properties and the impact this had on the phosphorus sink capacity of the soil. This experiment provides an evaluation of whether the life expectancy model developed previously by Baker et al. (2010) provides a useful estimation of vegetative treatment area phosphorus saturation life and explores what mechanisms may be allowing further phosphorus accumulation.

Nitrogen Fate and Retention in Vegetative Treatment Areas for Feedlot Runoff Control

This section (chapters 8-9) consists of two manuscripts that utilize different monitoring procedures and assays to assess mechanisms nitrogen fate and cycling within the vegetative treatment area. The first manuscript, "Vegetative treatment system impacts on groundwater quality," discusses groundwater concentrations up-gradient, within, and down-gradient of six vegetative treatment system on beef feedlots in Iowa. The manuscript provides statistical comparisons among average groundwater concentrations at each well and a trend analysis to evaluate impacts the system may be having. Nitrate leaching in the vegetative treatment system is also estimated. This article will be submitted to *Transactions of the ASABE*. The final manuscript, "The impact of vegetative treatment area use on soil biologically available carbon and nitrogen pools," reports the results of a long-term carbon and nitrogen fractionation procedure to evaluate if accumulation of labile carbon and nitrogen is occurring and if this organic matter is enriched. This article will be submitted to the *Soil Science Society of America Journal.*

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Chapter 2. Using Total Solids Concentration to Estimate Nutrient Content of Feedlot Runoff Effluent from Solid Settling Basins, Vegetative Infiltration Basins, and Vegetative Treatment Areas

Modified from a paper published in *Applied Engineering in Agriculture*

D.S. Andersen, R.T. Burns, L.B. Moody, M.J. Helmers

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Abstract. *Increased environmental awareness has promoted the need for improved feedlot runoff control. The use of vegetative treatment systems (VTSs) to control and treat feedlot runoff may enhance environmental security and protect water quality. Knowledge of effluent nutrient concentrations throughout the vegetative treatment system is required to evaluate system performance and impact on water quality. Previously collected VTS monitoring data have provided the opportunity to investigate relationships between effluent quality parameters. The objective of this study was to evaluate, through correlation and regression, the relationships between total solids, nutrients (ammoniacal nitrogen, Kjeldahl nitrogen, nitrate-nitrogen, total phosphorus, and orthophosphorus), and effluent quality indicator (five-day biological oxygen demand, chemical oxygen demand, chloride, and pH) concentrations of feedlot runoff at various stages of treatment in a VTS, including solid settling basin, vegetative infiltration basin, and vegetative treatment area effluent. Results of a correlation and primary factor analysis showed that most of the effluent concentrations were strongly correlated to each other, with a single factor capable of describing more than 60% of the total variability of the monitored parameters. Regression equations were developed to relate nutrient content and effluent quality indicator concentrations to total solids concentrations. Results were satisfactory (R^2 > 0.50) for ammoniacal nitrogen (NH_3-N), five-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), chloride (Cl^-),total* *phosphorus (TP), and total Kjeldahl nitrogen (TKN), indicating that total solids concentrations provided significant insight into VTS performance relative to nutrient concentration and effluent quality indicators. A comparison between predicted, based on total solids content, and monitored annual mass release of the parameters was conducted. No statistical difference was found for NH_3-N, BOD_5, COD, Cl^-, TP, and TKN; indicating that effluent volume release along with total solids concentrations could be used to provide an estimate of nutrient mass in solid settling basin, vegetative infiltration basin, and vegetative treatment area effluent.*

Keywords. *feedlot runoff, vegetative treatment systems, solid settling basin, vegetative treatment areas, vegetative infiltration basins, nutrient content, correlation, regression, total solids*

INTRODUCTION

Runoff from open-lot animal feeding operations (AFOs) has been recognized as a potential pollutant source to receiving waters because it contains nitrogen, phosphorus, organic matter, solids, and pathogens. The U.S. Environmental Protection Agency (EPA) developed a set of effluent limitation guidelines (ELGs) that described the design and operating criteria for feedlot runoff control systems on concentrated animal feeding operations (CAFOs) (Anschutz et al., 1979). These effluent limitation guidelines historically required collection, storage, and land application of feedlot runoff; however, recent modifications allowed the use of alternative treatment systems when the performance of the alternative systems, based on the mass of nutrients released, was equivalent to or exceeded that of an appropriately sized and managed containment system (EPA, 2006). One method of making this comparison was to use simulation models, along with site-specific climate and wastewater characterization data, to determine the pollutant discharge level that the alternative treatment and the containment basin systems would achieve (EPA, 2006).

Vegetative treatment systems (VTSs) are one possible alternative runoff control technology that has been proposed. A VTS is a combination of treatment components, at least one of which utilizes vegetation, to manage runoff from open lots (Koelsch et al., 2006). Vegetative treatment areas (VTAs) and vegetative infiltration basins (VIBs) are two possible treatment components for VTSs. A vegetative treatment area is a band of planted or indigenous vegetation situated down-slope of cropland or an animal production facility that provides localized erosion protection and contaminant reduction (Koelsch et al., 2006). As vegetative treatment technology has matured different types of treatment systems have been developed; for example, Bond et al. (2011) discuss costs associated with constructing sloped, level, pumped, and sprinkler vegetative treatment areas along with vegetative infiltration basins. Briefly, a sloped VTA is an area level in one dimension, to facilitate sheet flow, with a slight slope along the other, planted and managed to maintain a dense stand of perennial vegetation (Moody et al., 2006). Operation of a sloped VTA consists of applying solid settling basin effluent uniformly across the top of the vegetated treatment area and allowing the effluent to sheet-flow down the slope, whereas a level VTA uses a flood effect to distribute the effluent over the VTA surface. A pumped VTA has the increased flexibility of allowing the treatment area to be located upslope of the cropland or animal production facility, but still relies on flow to distribute effluent over the length of the vegetative treatment area surface. A sprinkler VTA has the same location flexibility as a pumped VTA, but has the additional advantage of uniform effluent application over the treatment area surface. Ikenberry and Mankin (2000) identified several possible methods in which effluent was treated by VTAs, including settling solids, infiltrating the runoff, and filtering of the effluent as it flowed through the vegetation. Additionally, interactions between soil and soil fauna and the flowing effluent could provide additional mechanisms of nutrient retention. A VIB is a flat area, surrounded by berms, planted to permanent vegetation. A VIB uses a flood effect to distribute effluent over the surface. These areas have drainage tiles located 1 to 1.2 m (3.4 to 4 ft) below the soil surface to encourage infiltration of effluent. The tile lines collect effluent that percolates through the soil profile. The effluent then receives additional treatment, often through the use of a VTA. Nutrient and pathogen removal in the VIB relies on effluent filtration as it percolates through the soil, plant uptake and removal through harvest, microbial degradation of the nutrients and pathogens by soil fauna, and sorption of contaminants to soil particles. Although these processes provide some treatment, nutrient and contaminant movement is still possible by convective transport if the parameter is dissolved or via colloid facilitated transport for particulate substances.

Young et al. (1980) and Dickey and Vanderholm (1981) provided two of the earlier studies of vegetative treatment of feedlot runoff. In their study Young et al. (1980) found that concentrations of total phosphorus, ortho-phosphorus, total Kjeldahl nitrogen, and ammonium nitrogen all decreased linearly down the length of the vegetative treatment area and found that percent reductions in total solids transported were similar to those for total phosphorus. Similarly, Dickey and Vanderholm (1981) found that concentrations of total Kjeldahl nitrogen, ammonia nitrogen, and chemical oxygen demand all showed similar reduction patterns as total solids down the length a vegetative treatment area. Dillaha et al. (1988) suggested that vegetative filtration changes flow hydraulics enhancing the opportunity for sedimentation of solids. More recent applications of vegetative treatment systems have been reported by Woodbury et al. (2003) and Faulkner et al (2011 a & b). Woodbury et al.

(2003) used a solid settling basin – sloped vegetative treatment area system to control and treat runoff from a beef feedlot in Nebraska. Over a three year monitoring period no release from the vegetative treatment area were reported. Faulkner et al. (2011a) reported on the use of a vegetative treatment area system for controlling silage bunker runoff. The Faulkner et al. (2011b) site was underlain by a shallow fragipan that restricted drainage and limited impacts on deep groundwater, but also contributed to surface flow releases.

These studies, along with the review of (Koelsch et al., 2006), have shown that vegetative treatment systems can be successful in a variety of situations. This has led to increased interest in their use on animal feeding operations for control of various wastewaters. As part of the permitting process on CAFO sized operations EPA requires modeling the performance of the proposed control system and suggests the use of site-specific wastewater characterization data. Recent research (Andersen et al., 2009) has shown that effluent concentrations from runoff control systems components can vary substantially from site to site, thus the use of book-values (American Society Agricultural and Biological Engineers, Midwest Plan Service) to predict nutrient concentrations could be highly inaccurate. Likewise, Edwards et al. (1986) reported high year-to-year variation in effluent concentrations with annual averages varying by approximately a factor of two for effluent from the feedlot, settling basin, and infiltration basin for total solids, chemical oxygen demand, nitrate, ammonia, organic nitrogen, total phosphorus, and soluble phosphorus. Moreover, numerous authors (Swanson et al., 1971; Swanson and Mielke, 1973; Andersen et al., 2009) have shown that event-to-event variability in feedlot runoff and solid settling basin effluent concentrations can be quite large. This it isn’t unexpected as event-to-event variability in storm pattern, size, and feedlot surface characteristics can be substantial, which can lead to large variations in runoff hydrology. This suggests that the use of book-values may not be sufficient for modeling control system performance.

Moreover, CAFOs utilizing alternative treatment systems are required to monitor system performance to ensure that the system is meeting minimum performance standards. Chemical analysis in the laboratory could provide high accuracy, but is expensive in terms of both the time and resources required to collect effluent samples and to carry out the laboratory analysis. Moreover, the results from the chemical analysis are often provided several weeks after sample collection; this limits applicability for making real-time decisions and other practical applications, particularly, since manure composition can change with time. This has led to interest in developing rapid methods for estimating nutrient concentrations of animal manures based on physicochemical properties. Previous studies (Chen et al., 2008; Marino et al., 2008; Moral et al., 2005) have attempted to relate manure slurry nutrient content to easily measured parameters including pH, total solids content, and electrical conductivity using linear regression and artificial neural network modeling. These studies have met with varying degrees of success, often finding that such relations are species and sometimes region dependent. For instance, Chen et al. (2008) investigated the use of multiple linear regression, polynomial regression, and artificial neural networks to model the nutrient concentrations of dairy manures finding that the artificial neural network model was most successful in estimating nutrient concentrations on dairies in China. Moral et al. (2005) evaluated the potential of linear relationships among nutrient contents and other easily measured parameters on pig slurries in Southeast Spain, finding that electrical conductivity was a strong predictor of ammoniacal nitrogen and potassium concentrations. Marion et al. (2008) suggested that dry matter content and electrical conductivity were good predictors of variables of agronomic interest for liquid dairy manures. In another study Kim and Gilley (2008) applied artificial neural network modeling to estimate erosion and nutrient concentrations in runoff from manure land application areas. In this study manure was surface applied once and then a rainfall simulator was used to create runoff 4, 32, 62, 123, and 354 days following manure application.

Gilley et al. (2009) found that concentrations of particulate phosphorus, ammonium-nitrogen, nitrate-nitrogen, and electrical conductivity were significantly correlated to feedlot soil characteristics. Based on this, Gilley et al. (2008) suggested that it may be possible to predict runoff nutrient concentrations based on the electrical conductivity of the feedlot soil (which serves as an indicator of soil dissolved solids). If, as Gilley et al. (2009) suggest, nutrient concentrations in feedlot runoff effluent were significantly related to feedlot soil characteristics, and as shown by Chen et al. (2008), Marino et al. (2008), and Moral et al. (2005) that nutrient content of manures is often related to solids content, then we hypothesize that there would be a strong correlation between the total solids concentration and nutrient content in feedlot runoff and total solids could potentially be used as an estimator of other water quality parameters.

This estimation method could serve several purposes; first, it has the potential to be used to better evaluate the impact feedlot runoff could be having on water quality. This information could be useful for prioritizing sites in need of enhanced or improved runoff control systems. For instance, Baker (2005) developed a model to assess the impact a feedlot would have on surface waters. Relating nutrient concentrations to total solids could provide improvements to models of this type by providing a simple mechanism by which nutrient concentrations could be modeled. Second, at many locations feedlot runoff is land applied as a nutrient source for crops. The estimation method could be used to provide an estimate of the appropriate application rate required to meet crop nutrient demand. The effluent could be tested for solids just prior to the application event and the nutrient estimate used to determine the application rate, Third, CAFOs utilizing vegetative treatment systems are required to perform substantial monitoring to validate the performance of their runoff control system; moreover, this data can be useful in making system management decisions and in determining appropriate system modifications. This monitoring can be expensive as every VTS release event needs to be sampled for numerous nutrient and effluent quality indicators. An estimation method has the potential to reduce these costs by allowing an estimate of nutrient mass release to be calculated based on fewer, more-easily monitored parameters. Additionally, the sample handling and preservation strategies required for certain parameters, such as total solids, are much less stringent than those required for nutrients and could thus reduce the effort required in sampling. Thus the opportunity to utilize an indicator parameter offers the opportunity to make more timely management decisions and to reduce time required in preparing samples for shipment for analysis. The estimation method could also be utilized to approximate nutrient content of the feedlot effluent throughout treatment, providing a better indication of how the runoff control system is performing and offering the operator with opportunity to improve system management. Finally, relating nutrient retention to sediment capture offers the potential to perform detailed modeling on the solids in the runoff and then use this as a proxy to understand nutrient reductions. This methodology has the potential to allow development of algorithms that would provide a more detailed description of how treatment is occurring within the runoff control system, leading to optimized system designs.

The objective of this study was to evaluate the use of total solids concentrations of effluent at various stages of treatment (at solid settling basin, vegetative infiltration basin, and vegetative treatment area outlets) to predict nutrient (ammoniacal nitrogen, Kjeldahl nitrogen, nitrate-nitrogen, total phosphorus, and orthophosphorus) and effluent quality indicator (five-day biological oxygen demand, chemical oxygen demand, chloride, and pH) concentrations of feedlot runoff from solid settling basins and vegetative treatment components. This was conducted by performing correlation and regression analysis for effluent concentration samples collected on six Iowa sites over a four year period. Prediction equation verification was performed by evaluating the developed regression equations ability to predict nutrient concentrations on a validation data set and by comparing annual mass releases from each VTS component to the estimated nutrient mass release based on effluent total solids concentration.

MATERIALS AND METHODS

The performance of six vegetative treatment systems was monitored. These treatment systems were located on CAFO beef feedlots throughout the state of Iowa. At many of the locations more than one VTS was installed. At each site, one VTS was monitored by Iowa State University (ISU). Table 1 shows the VTS configuration, the number of head of cattle, and the areas of the feedlot (and additional drainage area if present), VIB (where applicable), and VTA for the ISU-monitored systems. Full descriptions of these sites are available in Andersen et al. (2009).

Two different VTS configurations were monitored. These were a solid settling basin (SSB) followed by a VTA (SSB-VTA), and an SSB followed by a VIB in series with a VTA (SSB-VIB-VTA). In the SSB-VTA systems, runoff was collected from the beef feedlot and temporarily stored in a solid settling basin. Effluent from the solid settling basin was then released to the VTA. The VTA utilized gravity flow to spread the effluent down the length of the VTA. In the SSB-VIB-VTA systems, a solid settling basin captured the feedlot runoff. Solid settling basin effluent was released onto the VIB, and tile lines located 1 m below the VIB surface collected effluent draining through the VIB soil profile. This effluent was pumped onto a VTA for further treatment.

Table 1. Description of VTSs monitored by ISU including number of head, VTS configuration, and size of the feedlot, settling basin (SSB), vegetative infiltration basin (VIB), and vegetative treatment area (VTA).

Site	No. of Head	System Configuration	Feedlot Area (ha)	SSB Volume (m^3)	VIB Area (ha)	VTA Area (ha)
CN IA 1	1,000	1 SSB - 1 VTA	3.09	4,300	NA	1.52
CN IA 2	650	1 SSB - 1 VIB - 1 VTA	1.07	560	0.32	0.20
NW IA 1	1,400	1 SSB - 1 VTA	2.91	3,700	NA	1.68
NW IA 2	4,000	1 SSB - 1 VIB - 1 VTA	2.96	1,120	1.01	0.60
SW IA 1	2,300	1 SSB - 1 VTA	7.49	11,550	NA	4.05
SW IA 2	1,200	1 SSB - 1 VTA	3.72	6,300	NA	3.44

Monitoring Methods

Descriptions of the monitoring methodologies can be found in Moody et al. (2006) and Andersen et al. (2009). Briefly, Isco samplers (6712 portable samplers, Teledyne Isco, Lincoln, Neb.) were equipped with either a pressure transducer (720 submerged probe module, Teledyne Isco, Lincoln, Neb.) or an area-velocity meter (750 area velocity module, Teledyne Isco, Lincoln, Neb) and programmed with site and VTS component specific programs that collected multiple samples from each runoff event based on cumulative flow volumes. One sample, believed to be closest to the peak of the hydrograph, was selected for analysis per flow event. The sample was determined by noting sample collection times and the volume of flow programmed to occur between samples and determining an approximate hydrograph. After collection, the samples were placed on ice and shipped to a certified laboratory for analysis following chain-of-custody protocol during sample shipment. Effluent samples were analyzed for ammoniacal-nitrogen (NH_3-N), five-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), chloride (Cl^-), pH, total phosphorus (TP), total dissolved solids (TDS), total Kjeldahl nitrogen (TKN), total suspended solids (TSS), nitrate-nitrogen (NO_3-N), ortho-phosphorus (OP), and Fecal Coliform (FC) concentrations. Total solids (TS) content was calculated as the sum of TDS and TSS.

Data Analysis

For this study, all concentration data, except pH, were log transformed prior to statistical analysis to correct for normality (normality was tested using the Shapiro-Wilk test). Pearson correlation and regression analysis were conducted to determine correlation among sampled parameters and to find equations to predict nutrient/contaminant concentrations. Correlation analysis was performed on the entire data set using the PROC CORR command in SAS 9.2. A separate correlation analysis was performed for each VTS component, i.e., the SSB, VIB, and VTA. A primary factor analysis was conducted in SAS 9.2 using the PROC FACTOR command. A factor analysis is a statistical method used to describe variability among observed variable in terms of a potentially lower number of unobserved variables, called factors. In this analysis it was used to determine how many variables were required to describe the variability of the dataset.

A regression analysis was then conducted. The data set for each VTS component was randomly divided into calibration and validation data sets (1/2 of dataset used in calibration, 1/2 used in validation by assigning each sample a random number using Microsoft Excel, sorting the samples in ascending order, and utilizing the first half of the data, with the lowest random numbers, as the calibration data set). The data from all sites were pooled together for each treatment component before dividing the data sets. A linear regression analysis, on the log values of the concentration data, was performed in Microsoft Excel on the calibration data set to generate relationships between the variable of interest and the total solids concentration. The regression equations were then applied to the validation data. Modeling statistics and graphical comparisons were used to determine the ability of the developed regression equations to predict effluent concentrations. Modeling statistics used were the Nash-Sutcliffe efficiency (NSE), percent bias (BIAS), and the ratio of the root mean square error to the standard deviation of the monitored results (RSR). The NSE provided a measure of how well the predicted values followed the trends of the monitored data, BIAS measured the average tendency of the predicted data as compared to the monitored data, and RSR provided an index to evaluate the magnitude of the residual variations (Moriasi et al., 2007).

In addition to the above analysis, the prediction intervals were determined for each of the regression equations developed. The prediction interval provides a confidence interval on future observed responses, thus it provides an indication of how well the prediction equation works and the certainty with which the prediction can be made. It provides the net accuracy of the regression equation, as it states the 90% confidence interval around the mean of the selected value.

RESULTS AND DISCUSSION

Correlation Analysis

Correlation tests the extent to which two variables are linearly related. Pearson correlation coefficients among the tested parameters for the SSB, VIB, and VTA effluent were determined. Results were similar for all three components and are shown in Table 2, Table 3, and Table 4 for the SSB, VIB, and VTA respectively. We defined a strong correlation as having a value of 0.7 or more, as this would indicate that 50% of the variability of the parameters was shared. Based on this interpretation, many of the parameters were strongly correlated to each other, with only pH, nitrate, and fecal coliforms showing no strong correlations to the other parameters. Due to the correlation among the variables, a factor analysis was performed to assess how much of the variability was due to common factors, i.e., the communality of the dataset. The factor analysis of the settling basin effluent indicated that a single factor could explain 62% of the total variability for the effluent quality parameters. No additional factor could explain more than 9% of the dataset's variability. This indicated that only a single variable was justified in the regression equations. Factor analysis was also conducted for the VIB and VTA effluent. Results indicated that a single factor could again explain 61% and 68% of the total variability, with no other factors explaining more than 13% and 10% of the total variability, respectively. Based on the primary factor analysis, four parameters (total solids, total dissolved solids, total Kjeldahl nitrogen, and chemical oxygen demand) were strongly correlated to the primary factor. Total solids concentration was selected for use in the regression analysis as it is an easily measured parameter and it has the possibility to provide insight into transport of both particulate and dissolved parameters in that it is composed of both a dissolved and particulate components. It has the potential to track treatments through sedimentation, interaction with soil particles, and dilution from outside water sources (rainfall, run-on, etc.) as solids are affected by all three treatment processes.

Table 2. Pearson correlation coefficients for effluent from the solid settling basin[a]. Values in bold are statistically significant.

	NH_3	BOD_5	COD	Cl	pH	TP	TKN	TSS	NO_3	OP	TDS	TS
BOD_5	**0.80**	---	---	---	---	---	---	---	---	---	---	---
COD	**0.81**	**0.90**	---	---	---	---	---	---	---	---	---	---
Cl	**0.58**	**0.54**	**0.63**	---	---	---	---	---	---	---	---	---
pH	**-0.54**	**-0.58**	**-0.59**	**-0.34**	---	---	---	---	---	---	---	---
TP	**0.79**	**0.77**	**0.84**	**0.54**	**-0.56**	---	---	---	---	---	---	---
TKN	**0.86**	**0.86**	**0.92**	**0.64**	**-0.54**	**0.82**	---	---	---	---	---	---
TSS	**0.62**	**0.72**	**0.79**	**0.52**	**-0.45**	**0.70**	**0.77**	---	---	---	---	---
NO_3	0.08	0.07	**0.15**	**0.21**	-0.08	**0.15**	**0.15**	**0.15**	---	---	---	---
OP	**0.62**	**0.57**	**0.58**	**0.37**	**-0.52**	**0.78**	**0.57**	**0.37**	**0.18**	---	---	---
TDS	**0.79**	**0.79**	**0.86**	**0.76**	**-0.52**	**0.77**	**0.86**	**0.74**	**0.21**	**0.53**	---	---
TS	**0.75**	**0.79**	**0.89**	**0.72**	**-0.52**	**0.80**	**0.87**	---	**0.20**	**0.50**	---	---
FC	0.05	**0.21**	**0.22**	**0.17**	**-0.21**	0.08	**0.17**	**0.32**	**0.11**	-0.07	**0.22**	**0.26**

[a] A correlation coefficient is significant at the 95% confidence level if |correlation| > 0.11. Data represent 434 samples.

Table 3. Pearson correlation coefficients for effluent from the vegetative infiltration basin[b]. Values in bold are statistically significant

	NH_3	BOD_5	COD	Cl	pH	TP	TKN	TSS	NO_3	OP	TDS	TS
BOD_5	**0.84**	---	---	---	---	---	---	---	---	---	---	---
COD	**0.86**	**0.95**	---	---	---	---	---	---	---	---	---	---
Cl	**0.68**	**0.60**	**0.60**	---	---	---	---	---	---	---	---	---
pH	0.12	0.05	0.06	**0.15**	---	---	---	---	---	---	---	---
TP	**0.80**	**0.90**	**0.92**	**0.57**	0.07	---	---	---	---	---	---	---
TKN	**0.88**	**0.92**	**0.95**	**0.63**	0.08	**0.91**	---	---	---	---	---	---
TSS	**0.39**	**0.61**	**0.60**	**0.26**	**-0.20**	**0.67**	**0.57**	---	---	---	---	---
NO_3	**-0.14**	0.08	0.07	-0.07	-0.04	**0.15**	0.08	**0.32**	---	---	---	---
OP	**0.67**	**0.80**	**0.80**	**0.48**	**0.16**	**0.85**	**0.76**	**0.58**	**0.31**	---	---	---
TDS	**0.83**	**0.83**	**0.83**	**0.66**	**0.16**	**0.76**	**0.83**	**0.43**	-0.01	**0.65**	---	---
TS	**0.66**	**0.79**	**0.78**	**0.50**	-0.07	**0.81**	**0.78**	---	**0.17**	**0.67**	---	---
FC	**0.39**	**0.64**	**0.63**	**0.25**	-0.03	**0.59**	**0.61**	**0.43**	**0.28**	**0.54**	**0.49**	**0.51**

[b] A correlation coefficient is significant at the 95% confidence level if |correlation| > 0.13. Data represent 237 samples.

Table 4. Pearson correlation coefficients for effluent from the vegetative treatment basin[c]. Values in bold are statistically significant

	NH_3	BOD_5	COD	Cl	pH	TP	TKN	TSS	NO_3	OP	TDS	TS
BOD_5	**0.90**	---	---	---	---	---	---	---	---	---	---	---
COD	**0.92**	**0.96**	---	---	---	---	---	---	---	---	---	---
Cl	**0.64**	**0.64**	**0.71**	---	---	---	---	---	---	---	---	---
pH	**-0.48**	**-0.50**	**-0.47**	-0.03	---	---	---	---	---	---	---	---
TP	**0.87**	**0.83**	**0.88**	**0.60**	**-0.48**	---	---	---	---	---	---	---
TKN	**0.95**	**0.93**	**0.96**	**0.71**	**-0.47**	**0.89**	---	---	---	---	---	---
TSS	**0.78**	**0.85**	**0.86**	**0.55**	**-0.48**	**0.74**	**0.83**	---	---	---	---	---
NO_3	0.07	0.08	0.05	-0.02	-0.13	0.13	0.09	0.01	---	---	---	---
OP	**0.81**	**0.73**	**0.77**	**0.54**	**-0.44**	**0.91**	**0.80**	**0.59**	**0.16**	---	---	---
TDS	**0.80**	**0.86**	**0.90**	**0.84**	**-0.27**	**0.75**	**0.86**	**0.76**	-0.01	**0.65**	---	---
TS	**0.81**	**0.87**	**0.91**	**0.80**	**-0.30**	**0.75**	---	**0.86**	-0.01	**0.63**	---	---
FC	**0.47**	**0.54**	**0.54**	**0.28**	**-0.42**	**0.51**	**0.51**	**0.55**	0.06	**0.45**	**0.45**	**0.47**

[c] A correlation coefficient is significant at the 95% confidence level if |correlation| > 0.13. Data represent 229 samples.

Regression Equation Calibration

Linear regression was performed on the log of the concentration data to relate parameter concentration to total solids concentration for the SSB, VIB, and VTA effluent. Developed regression equations are shown in Table 5. The amount of the variability described by the regression equation is also provided (R^2). Several parameters, (pH, NO_3-N, ortho-phosphorus, and fecal coliform) could not be described by the regression equations as indicated by the low (less than 0.50) R^2 values. In addition, the 90% prediction interval is also provided for each equation. The prediction interval provides a confidence interval on future observed responses.

Table 5. Regression equations relating solid settling basin (SSB), vegetative infiltration basin (VIB), and vegetative treatment area (VTA) effluent contaminant concentrations to total solids concentrations. The R^2 valve of each regression equation is provided. PI is the 90% prediction interval, i.e., 90% of future measurements of the dependent variable fall inside the interval.

Dependent Variable	SSB			VIB			VTA		
	Regression Equation	R^2	90% PI	Regression Equation	R^2	90% PI	Regression Equation	R^2	90% PI
NH_3-N	$=1.42*10^{-2}(TS)^{1.00}$	0.56	$=10^{\log y\pm0.54}$	$=5.56*10^{-4}(TS)^{1.27}$	0.39	$=10^{\log y\pm0.93}$	$=1.43*10^{-5}(TS)^{1.81}$	0.66	$=10^{\log y\pm0.92}$
BOD_5	$=1.74*10^{-2}\ (TS)^{1.24}$	0.61	$=10^{\log y\pm0.60}$	$=9.57*10^{-6}\ (TS)^{2.02}$	0.60	$=10^{\log y\pm0.96}$	$=4.52*10^{-5}\ (TS)^{1.95}$	0.78	$=10^{\log y\pm0.75}$
COD	$=2.77*10^{-1}(TS)^{1.10}$	0.76	$=10^{\log y\pm0.37}$	$=5.13*10^{-3}\ (TS)^{1.48}$	0.62	$=10^{\log y\pm0.67}$	$=6.27*10^{-3}\ (TS)^{1.53}$	0.84	$=10^{\log y\pm0.47}$
Cl^-	$=1.24(TS)^{0.65}$	0.52	$=10^{\log y\pm0.38}$	$=8.87(TS)^{0.42}$	0.30	$=10^{\log y\pm0.38}$	$=2.62*10^{-1}\ (TS)^{0.83}$	0.62	$=10^{\log y\pm0.46}$
pH	$=9.68-0.62\log(TS)$	0.33	$=pH\pm0.53$	$=7.40-0.11\log(TS)$	0.01	$=pH\pm0.59$	$=8.78-0.67\log(TS)$	0.16	$=pH\pm0.60$
TP	$=1.58*10^{-1}\ (TS)^{0.69}$	0.62	$=10^{\log y\pm0.33}$	$=4.08*10^{-4}(TS)^{1.29}$	0.65	$=10^{\log y\pm0.55}$	$=1.05*10^{-2}\ (TS)^{0.97}$	0.61	$=10^{\log y\pm0.56}$
TKN	$=3.28*10^{-2}\ (TS)^{1.02}$	0.72	$=10^{\log y\pm0.39}$	$=5.98*10^{-4}\ (TS)^{1.41}$	0.61	$=10^{\log y\pm0.65}$	$=3.84*10^{-4}\ (TS)^{1.54}$	0.76	$=10^{\log y\pm0.61}$
NO_3-N	$=2.92*10^{-1}\ (TS)^{0.17}$	0.02	$=10^{\log y\pm0.74}$	$=8.13*10^{-2}(TS)^{0.33}$	0.03	$=10^{\log y\pm1.18}$	$=6.50*10^{-1}\ (TS)^{0.07}$	0.00	$=10^{\log y\pm0.80}$
OP	$=6.08*10^{-1}\ (TS)^{0.48}$	0.25	$=10^{\log y\pm0.51}$	$=6.25*10^{-6}\ (TS)^{1.59}$	0.37	$=10^{\log y\pm1.20}$	$=2.37*10^{-2}\ (TS)^{0.80}$	0.39	$=10^{\log y\pm0.71}$
FC	$=744(TS)^{1.05}$	0.08	$=10^{\log y\pm2.19}$	$=9.52*10^{-6}\ (TS)^{2.82}$	0.26	$=10^{\log y\pm2.81}$	$=5.93*10^{-4}\ (TS)^{2.52}$	0.27	$=10^{\log y\pm2.93}$

Note: In the 90% PI *y* represents the dependent variable.

Regression Equation Validation

The regression equations' ability to predict constituent concentration based on the total solids concentrations in the SSB, VIB, and VTA effluent was then tested. This testing used the validation data set. Figure 1 shows the ability of the regression equations, based on TS concentrations, to predict parameter concentrations for NH_3-N, TKN, TP, and COD. The calibration equations were also evaluated with the use of modeling statistics. The modeling statistics used were the NSE, BIAS, and the RSR. Modeling statistics results are provided in Table 6.

Table 6. The Nash-Sutcliffe efficiency (NSE), percent bias (BIAS), and the ratio of the root mean square error to the standard deviation of the monitored results (RSR) for evaluating regression equation performance for ammoniacal-nitrogen, five-day biochemical oxygen demand, chemical oxygen demand, chloride, pH, total phosphorus, total Kjeldahl nitrogen, nitrate-nitrogen, orthophosphorus, and fecal coliform.

	SSB			VIB			VTA		
	NSE	RSR	BIAS	NSE	RSR	BIAS	NSE	RSR	BIAS
NH_3	0.61	0.62	26	0.15	0.92	62	0.68	0.56	20
BOD_5	0.67	0.57	28	-0.07	1.03	36	0.57	0.65	20
COD	0.75	0.50	19	0.32	0.82	37	0.74	0.51	16
Cl^-	0.38	0.78	10	0.27	0.85	10	0.61	0.62	18
pH	0.21	0.89	0	-0.02	1.01	-1	0.00	1.00	0
TP	0.67	0.58	12	0.49	0.71	22	0.55	0.67	24
TKN	0.76	0.49	17	0.26	0.86	32	0.80	0.44	10
NO_3-N	-0.07	1.03	39	-0.19	1.09	71	-0.09	1.04	58
OP	0.28	0.85	19	0.10	0.95	64	0.26	0.85	36
FC	-0.06	1.03	95	-0.02	1.01	93	-0.04	1.02	98
Ideal Value	**1.00**	**0.00**	**0**	**1.00**	**0.00**	**0**	**1.00**	**0.00**	**0**

All regression equations were found to have a tendency to underestimate parameter concentrations as evidenced by the positive value for the BIAS statistic. The NSE provided information about the regression equations' ability to follow trends in concentration, with values greater than zero indicating that the regression equation performs better than using the average of the monitored data; for all parameters except pH, NO_3-N, and fecal coliforms the regression equations provided a better predictor than using the average value (positive NSE values). This indicates that use of these regression equations, rather than averages or table values, may provide a better estimate of parameter concentrations. The RSR value compared the standard deviation of the monitored results to the residual variability remaining after applying the regression equation; values less than one indicated that the regression equation described more variability than the mean value of the monitored data. It appeared that many of the regression equations were providing a good description of the parameter concentrations, indicating that total solids concentration had the potential to serve as a proxy for better understanding the treatment, in terms of the nutrient concentrations reduction that VTSs are achieving.

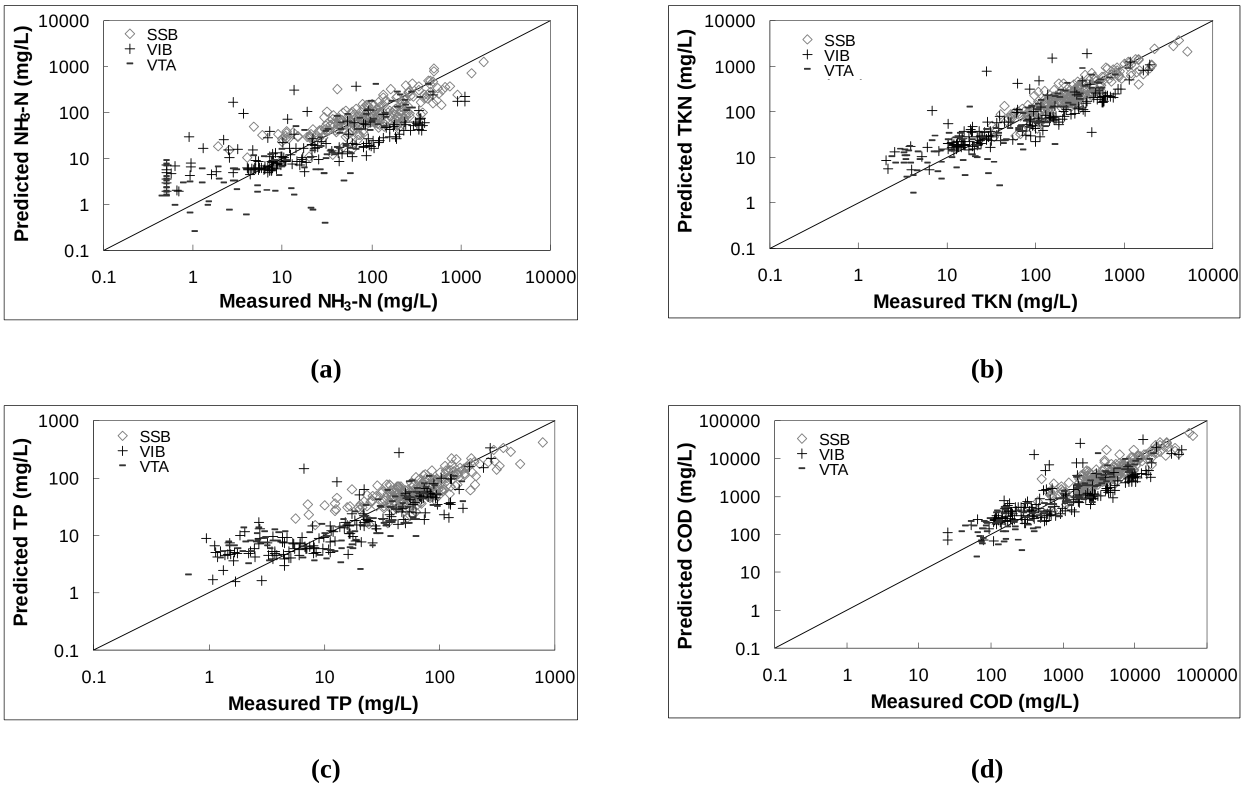

Figure 1. Plots of predicted, based on TS concentrations, versus modeled (a) ammoniacal-nitrogen (NH_3-N), (b) total Kjeldahl nitrogen (TKN), (c) total phosphorus (TP), and (d) chemical oxygen demand (COD) concentrations for solid settling basin (SSB), vegetative infiltration basin (VIB), and vegetative treatment area (VTA) effluent. The one-to-one line is also displayed in the graphs.

IMPLICATIONS

The introduction discussed five potential uses for a nutrient/contaminant concentration estimation methodology. These included using total solids concentrations to evaluate the impact feedlot runoff was having on water quality, using total solids as a proxy to determine effluent application rates for use as a fertilizer or in determining nutrient loading rates on vegetative infiltration basins and vegetative treatment areas, as part of monitoring the VTS releases as required in NPDES permits issued to animal feeding operations, making timely management systems involved in operating VTS and evaluating overall system performance, and in developing detailed process based algorithms to describe nutrient retention in vegetative treatment systems. This section of the manuscript will provide examples to illustrate these potential applications and discuss how the proposed methodology offers potential for better modeling runoff control system performance.

In practice, determining effluent application rates for use as a fertilizer, loading rates on vegetative infiltration basins and vegetative treatment areas, and monitoring VTS releases are all essentially the same. In all three cases we are most interested in estimating yearly nutrient loadings rates or contaminant releases, that is, we want to estimate the mass of contaminant either in the effluent released from the system, applied to cropland, or retained within each treatment component. To test the use of these proposed regression equations for these purposes we compared the monitored annual contaminant mass transport and the annual contaminant mass transport estimated based on total solids concentrations. These evaluations were made for NH_3-N, BOD_5, COD, Cl^-, TP, and TKN. Evaluations for NO_3-N, and OP were not performed as the R^2 values of the regression equations indicated weak relationships. The monitored total solids concentration from each release event for each VTS component was used in the regression equation to project effluent concentrations. The estimated concentrations were multiplied by the event flow volume to determine mass release. Event mass releases were then summed to calculate the annual mass release. These calculated values were compared to the monitored mass release from each VTS component. A paired t-test was performed to determine if there was a statistical difference between the monitored and predicted mass release (Table 7). Significant differences in mass release estimates were only seen for NO_3-N and OP. These results indicated that this methodology offered considerable insight into determining appropriate effluent application rates for use as a fertilizer, evaluating contaminant masses released from the runoff control system, and in estimating nutrient loading rates onto the vegetative treatment system components.

Table 7. P-values for a paired t-test comparing monitored mass release to predicted mass release calculated based on total solids concentration. Significant differences are shown in italics.

Component	NH_3-N	BOD_5	COD	Cl^-	Total P	TKN
SSB	0.86	0.69	0.43	0.85	0.70	1.00
VIB	0.19	0.40	0.13	0.59	0.16	0.19
VTA	0.38	0.39	0.48	0.32	0.19	0.67

Likewise, evaluating the impact releases from a feedlot's runoff control system are having on water quality and developing detailed process-based algorithms to describe nutrient retention in vegetative treatment are similar tasks. In both cases, the proposed methodology regression equations would suggest that focusing on the transport of solids would provide a computationally efficient means of evaluating the systems performance relative to other nutrients. Recent work (Flanagan and Nearing, 2000 and Gao et al., 2004) has alluded to improving methodologies for quantifying transport of soil particles and dissolved solids in agricultural settings. It's possible that the models proposed in these manuscripts could be used to estimate solids transport from the feedlot surface. Hydraulic models and flow detention techniques could then be used to estimate solid settling within the basin and estimate solids concentrations at the outlet. The proposed regression equations could then be utilized to estimate nutrient concentrations of the effluent. This methodology offers a significant advantage over utilizing book-values as it compensates for both event-to-event variability in nutrient concentrations in runoff from a single lot and has the potential to characterize the risks that feedlots of various sizes (i.e., slope lengths), slope angles, and slope profiles would pose. Similarly, further sediment deposition and filtration that occurs in vegetative treatment areas and vegetative infiltration basins could be modeled and used as a proxy to model nutrient retention.

CONCLUSIONS

Feedlot runoff is receiving increased attention as a potential environmental contaminant. As a result, feedlots are seeking information on runoff control practices that enhance environmental security. Vegetative treatment systems are one option seeing increased use; however, knowledge of effluent nutrient concentrations throughout the treatment system is required to evaluate system performance and to make real-time management decisions. The objective of this research was to evaluate the use of total solids concentrations to predict nutrient concentrations of feedlot runoff undergoing vegetative treatment. This was done by performing a correlation and regression analysis. Results of the correlation analysis indicated that most of the parameter concentrations were significantly related to each other, with all parameters exhibiting a significant correlation with at least one other monitored parameter. A primary factor analysis showed a single factor was capable of describing more than 60% of the variability of the ten monitored parameters. Regression equations were developed to relate nutrient content and effluent quality indicator concentrations to total solids concentrations. Results were satisfactory for most parameters, indicating that total solids concentrations provided significant insight into the performance, in terms of nutrient concentrations reductions, VTSs were achieving. The predicted and monitored annual mass releases were compared for NH_3-N, BOD_5, COD, Cl^-, TP, and TKN; NO_3-N and OP were not evaluated as the regression equations indicated only a weak relationship. No statistically significant differences in mass release were found. This indicates that monitoring of TS mass release may be adequate to predict these nutrient mass releases from the VTS.

Acknowledgements

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Chapter 3. A Review of Settling Characteristics of Solids in Runoff from Beef Feedlots

Abstract. *Feedlot runoff is a potential environmental contaminant. Primary treatment of feedlot runoff typically relies on solids removal, usually through sedimentation techniques. Rigorous engineering design of sedimentation systems should consider flows into, within, and out of the settling structure, and the settling characteristics of the solids in the runoff. This review summarizes available literature on settling characteristics, including settling velocities, particle densities, particle sizes, effluent viscosity, and solids concentration effects on settling of solids in feedlot runoff. Findings indicate that solids in runoff from concrete lots settle more slowly than those in earthen lots, primarily due to lower particles densities (~1.5 g/cm^3 for concrete, 2.0 g/cm^3) for earthen presumably due to not being mixed with the denser soil particle present on the earthen feedlot surface. Particles size distributions exhibited substantial variation, both between and within lots, which appears to be attributable to event hydrology. Literature evidence indicates that at solids concentrations exceeding 15,000 and 20,000 mg/L from earthen and concrete lot runoff could result in hindered settling, which causes reductions of settling velocity of 10% or more. Current settling basin design standards are then evaluated in light of the discussed settling characteristics of runoff solids. These results facilitate physically based modeling of sedimentation within runoff control systems and could potentially be used to improve design recommendation for settling basins on open lot operations.*

Keywords. *Feedlot runoff, settling, solid settling basin design, waste treatment, particle density, particle size, cattle manure, settling velocity*

INTRODUCTION

Concentrating cattle in feedlots has numerous advantages in productivity and quality control; however, it results in an increased potential for surface and groundwater pollution (Sweeten et al., 1990). Preventing potential problems from developing into real problems requires feedlot operators to be proactive in installing runoff control systems, managing solid manure, and maintaining the feedlot surface. Research has been performed on these topics since the early 1970's, with much of the work focusing on various methods of treating and handling feedlot runoff. These research studies have provided a substantial pool of data on the physical and chemical properties of lot runoff, the hydraulic properties of feedlot surfaces, and the performance of solid settling systems. The objective of this review is to consolidate the data and techniques necessary to model solid settling systems based on the physical characteristics of the runoff. Specifically, the objectives of this work were to (1) evaluate the "settleability" of solids in the runoff, (2) evaluate the characteristics of the waste stream that contribute to these settling properties, and (3) to evaluate these settling characteristics in terms of current settling basin design standards and recommendations.

PARTICLE SETTLING THEORY

Sediment is composed of many materials, including individual mineral particles, aggregates, organic material, and their associated chemicals. The properties of this sediment, i.e., its size, shape, density, surface charge, etc., affect its settling velocity and in turn its transport. In general, settling regimes can be classified into four types, (1) discrete particle settling, (2) flocculent settling, (3) hindered or zone settling, and (4) compression settling. The settling regime that occurs is dependent on the concentration of solids in the solution and the tendency of the particles to interact. Feedlot runoff can, and typically does, experience all four regimens; however, for typical design purposes the process is usually assumed to be dominated by Type I-discrete particle settling. This process can be modeled with Stokes Law (Eq. 1).

$$v_s = \frac{g\left(\rho_p - \rho_f\right)d_p^2}{18\mu} \tag{1}$$

In this equation v_s is the settling velocity of the particle (m/s), g is the acceleration due to gravity (m/s^2), ρ_p is the particle density (kg/m^3), ρ_f is the density of the fluid (kg/m^3), d_p is the diameter of the particle (m), and μ is the dynamic viscosity of the fluid (N-s/m^2). Typical design of a settling basin involves selecting a critical settling velocity (v_c) and then sizing the settling structure such that all particles with a settling velocity equal to or greater than the selected critical velocity will be captured within the basin; however, scientific justification for selecting a specific critical settling velocity is lacking. After the critical settling velocity is selected it can be related to the required surface area of the settling basin by dividing the flow rate by this critical velocity (as shown in Eq. 2).

$$A = \frac{Q}{v_c} \tag{2}$$

In this equation A is the required surface area of the settling basin (m^2), Q is the flow rate of effluent into/out (assuming steady state) of the settling basin (m^3/s), and v_c is the selected critical particle settling velocity (m/s). Thus, in theory settling basin design requires the selection of two constraints, flowrate, which typically is chosen based on a design storm, and the critical settling velocity;

however, in practice settling basin design is complicated by factors such as unsteady flow, dead zones within the settling structure, and changing fluid and particle properties.

PHYSICAL CHARACTERISTICS OF FEEDLOT RUNOFF

Settling velocity distributions provide a direct measurement of the property of interest, i.e., the retention time required to achieve sedimentation of a specified fraction of the transported material. The measurement of settling velocity distributions provides a fundamental, but empirical, approach to evaluating runoff settleability. As such, the use of the settling velocity distribution provides a sound method to evaluate settling basin performance and design; however, a more comprehensive understanding of the parameters that cause the settling velocity distribution are useful for detailed modeling and extrapolating data beyond their original situation. As this is the case, the approach in this manuscript will be to first look at the settling velocity distribution, and then to focus on the various properties of the solid particles and runoff effluent that gave the waste stream the observed settling properties.

Settling Velocities

The settling velocity distribution provides a relationship between settling time and the percentage of particles remaining in solution. Data of this type have been presented in numerous manuscripts (Gilbertson et al., 1972; Gilbertson and Nienaber, 1973; Moore et al., 1975; Gilbertson and Nienaber, 1978; Lott et al., 1994; Pepple et al., 2011); however, differences in sampling depths between studies makes it difficult to directly compare information. To remove this constraint all data were transformed to represent a one meter sampling depth; this was done by dividing the depth of one meter by the settling velocity (sampling depth for the study divided by sample collection time) and calculating the percent of total solids remaining in the supernatant at the sample time (supernatant concentration divided by original total solids concentration times one hundred). As was done by Lott et al. (1994) and Pepple et al. (2011) a regression model was fitted to the settling velocity distributions to facilitate comparisons. In their work, Lott et al. (1994) used a four-parameter hyperbolic equation to fit the data while Pepple et al. (2011) used a decaying exponential equation (based on that of Branch-Papa et al., 2006); these are shown as Eqs. 3 and 4, respectively. In Eq. 3, y represents the percentage of material settled, t the settling time, and a, b, c, and d are fitting parameters with little physical meaning. In Eq. 4, A represents the percent of total solids that are settleable, B represents a time constant related to the distribution of particle settling rates of the feedlot runoff solids, and C represents the percent of total solids that are non-settleable. The advantage of Eq. 4 is that the meaning of all three of the fitting parameters can be interpreted; however, since this equation has fewer fitting parameters it has a reduced flexibility to match the shape of the settling velocity distribution. In this review we chose to use the equation of Pepple et al. (2011) as it aided in interpretation while still providing a good quantitative description (R^2 >0.65 for all samples and higher than 0.85 for all samples other than Lott et al., 1994) of the settling velocity distribution.

$$y = a + \frac{b}{1 + dt} + ct \tag{3}$$

$$TS = A\exp(-Bt) + C \tag{4}$$

This analysis broke the settling velocity distributions into two categories, earthen lots and concrete lots. Studies that worked with the settling properties of cattle feces from confinement operations were included in the concrete runoff data as they represented unaltered properties of the manure solids, i.e., there was no mixing of manure solids with soil particles. Only two studies (Pepple et al., 2011; Gilbertson and Nienaber, 1978) have provided examples of typical of settling distributions for concrete lot runoff solids. The Gilbertson and Nienaber (1978) study reported two settling velocity distributions for beef manure, one sample was beef manure (8% total solids) collected from a 100-head housed feedlot and the other was the same manure that had been run through a 30 mesh (595 μm) screen prior to the settling test. The Pepple et al. (2011) samples were collected from multiple runoff events at three different concrete lots. The average and standard deviation of the Pepple et al. (2011) settling velocity distributions and the two Gilbertson and Nienaber (1978) settling velocity distributions are shown in Figure 1. The Gilbertson and Nienaber (1978) samples settled similarly to those from Pepple et al. (2011). The screened cattle feces sample was similar to slower settling concrete runoff sample in terms of settling velocity distribution. The regular cattle manure feces sample collected by Gilbertson and Nienaber (1978) settled very similarly to the quicker settling samples from Pepple et al. (2011). The fact that many of the runoff samples had similar settleability to a screened manure sample could indicate that smaller solid particles are selectively transported during runoff events (see discussion on particle size distribution). Alternatively, this could indicate that biological degradation of the manure solids, which would cause a decrease in particle size, occurred prior to the runoff event, reducing settlability. The non-screened manure sample collected by Gilbertson and Nienaber (1978) resembled good settling events from the Pepple et al. (2011) data. This could indicate that the feedlot surface had been recently cleaned prior to the runoff event and available particles were similar to those from fresh feces or alternatively that the runoff event was large enough to transport all particles on the feedlot surface, rather than just smaller, more easily suspended and transported particles.

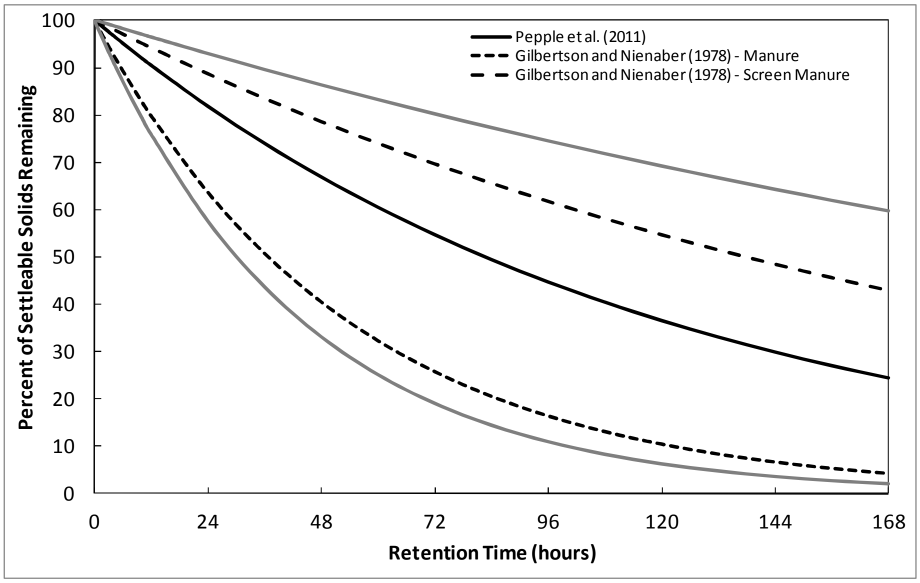

Figure 1. Settling velocity distributions for concrete lot runoff solids from Pepple et al. (2011) and settling velocity distributions of housed beef cattle feces from Gilbertson and Nienaber (1978). Graph is constructed for a 1 m deep settling basin. Solid grey lines represent average ± one standard deviation in settling rate from Pepple et al. (2011).

The same analysis was conducted for the earthen lot data. In this case data were available from Pepple et al. (2011), Gilbertson and Nienaber (1973), and Lott et al. (1994). The settling velocity distributions reported (Fig. 2) varied drastically among the three sources, with the settling velocities reported by Lott et al. (1994) being extremely quick and those of Pepple et al. (2011) being very slow in comparison. The velocity distributions reported by Gilbertson and Nienaber (1973) were slower than those of Lott et al. (1994), but still substantially faster than those of Pepple et al. (2010). Gilbertson and Nienaber (1973) and Pepple et al. (2011) collected samples from multiple runoff events and in the case of Pepple et al. (2011) from multiple feedlots. Lott et al. (1994) collected solids from the feedlot surface for particle size analysis. We hypothesize that differences in sample collection methodology (from the lot surface as opposed to from runoff events) and differences in hydrologic conditions between lots may have contributed to these differences. As the Lott et al. (1994) sample was collected from a feedlot surface the particle size distribution was not subject to the raindrop impacts that could break apart aggregates nor to the sorting of particles (due to selective erosion and transport) that would occur during runoff events; moreover, as this sample was collected during drier conditions. This led to distributions of solid particle sizes that had more large, quickly settling particles than those encountered in the natural rainfall events of Gilbertson and Nienaber (1973) and those of Pepple et al. (2011), possibly because rainfall had not broken up aggregates. Likewise, the Gilbertson and Nienaber (1973) feedlot was located on a much steeper slope (10%) than the lots sampled by Pepple et al. (0.5 – 5%). The gentler slope probably contributed to slower runoff rates and therefore more selective transport of smaller, more easily eroded and suspended particles. A similar hypothesis was put for by Miner et al. (1966) who stated physical transport on slightly sloping lots is limited to small particles while transport on steeper lots can be substantial. Additionally, the Pepple et al. (2011) study utilized grab sample collection while the Gilbertson and Nienaber study used electronic sampling equipment. Collection of the grab samples was delayed towards the later stages of the runoff event due to travel times to the lots, which may have resulted in lower sediment concentrations and finer particle sizes then those collected with the electronic equipment of Gilbertson and Nienaber where samples were collected near the peak of runoff hydrographs.

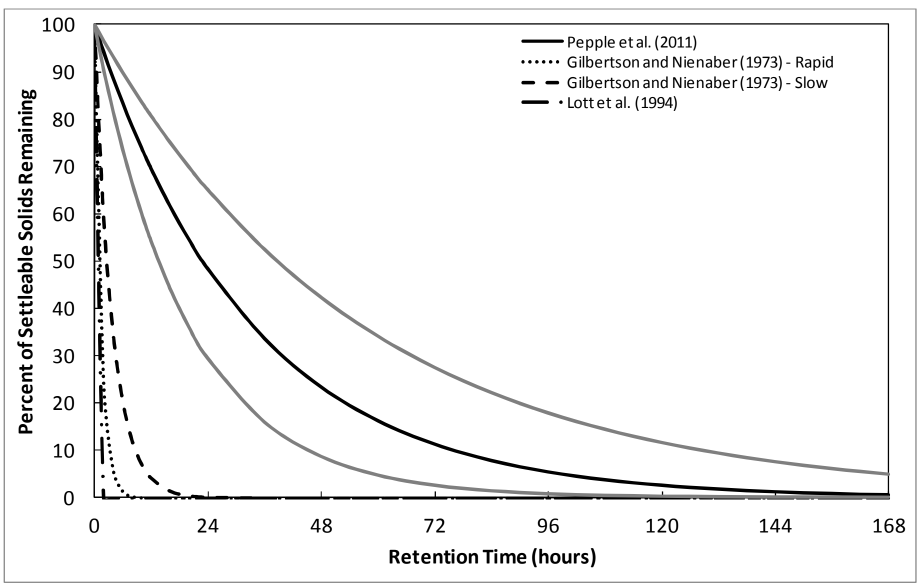

Figure 2. Settling velocity distributions for earthen lot runoff solids from Pepple et al. (2011), Gilbertson and Nienaber (1973), and Lott et al. (1994). Graph is constructed for a 1 m deep settling basin. Solid grey lines represent average ± one standard deviation in settling rate from Pepple et al. (2011).

Support of this hypothesis is provided by the data sets of Pepple et al. (2011) and Gilbertson and Nienaber (1973), both of which show substantial event-to-event variation in runoff settleability, with Gilbertson and Nienaber (1973) going so far as to classify certain events as good settling and others as poor settling. The fact that the variability occurs within a lot indicates that differences in event size and intensity, and therefore runoff rates, can have a large impact on sediment size distributions. This concept was investigated in rainfall simulator study of Gilley et al. (2011) in which they measured particle size distributions under various runoff rates, finding that median particle size was positively correlated with runoff rate. Differences in natural soil texture of the lots also may be an important factor in the settling velocity distribution as researchers have shown that between 25-50% of all solids removed during lot cleaning could be soil (Gilbertson et al., 1975; Parker et al., 2004).

Particle Densities

The density differences between the particles and fluid provide the gradient for settling, thus particle density is an important characteristic for determining the settleability of runoff solids. Currently four references (Gilbertson and Nienaber, 1973; Frecks and Gilbertson, 1974, Gilbertson et al., 1975; and Pepple et al., 2011) have reported particle densities of solids in feedlot runoff or of cattle manure. All four studies used a pycnometer procedure (Blake and Hartge, 1986) to determine particle density. Measured particle densities have been fairly consistent, with Pepple et al. (2011) and Gilbertson and Nienaber (1973) reporting average particle densities of 1.89 ± 0.11 g/cm^3 and 1.95 ± 0.18 g/cm^3, respectively, for solids in runoff from earthen surfaced feedlots. Gilbertson et al. (1975) also measured the particle density of solids obtained directly from the feedlot surface, finding an average particle density of 2.28 g/cm^3, which is substantially denser than those found to be transported in the lot runoff and may indicate preferential transport of the lighter particles. Pepple et al. (2011) reported the particle density of solids in concrete lot runoff to be 1.47 ± 0.17 g/cm^3; which is similar to the values Freck and Gilbertson (1973) reported for feces from cattle fed high roughage (1.53 ± 0.22 g/cm^3) and high concentrate (1.50 ± 0.23 g/cm^3) diets.

These results indicate several points; first diet does not appear to have a significant effect on particle density. Second, there appears to be a large difference in particle densities between earthen and concrete lots, with concrete lot particle densities being significantly lighter than earthen lot runoff solids. Researchers (Gilbertson et al., 1975; Parker et al., 2004) have reported that when cleaning earthen lots, a substantial amount of soil is removed with the cattle feces. Gilbertson et al. (1975) attributed the large amount of soil removed during feedlot cleaning to “animal mixing,” i.e., mixing of feedlot soil and animal feces due to the stirring action of the animal hooves, thus it is probable that the increased particle density of earthen lot runoff solids is due to manure and soil mixing as typically mineral particles have densities around 2.65 g/cm^3 (although there is substantial variation about this value depending on soil mineralogy and organic matter content). Regardless, both (earthen and concrete) particle densities are substantially lower than the 2.65 g/cm^3 often assumed for solid particle densities and used in feedlot runoff sedimentation models (Tolle et al., 2007); this can have a dramatic impact on settling rates and settling basin performance. For instance, earthen lot particles with a density of 1.92 g/cm^3 (the average of values from Pepple et al., 2011 and Gilbertson and Nienaber, 1973) would settle 44% slower than particles having a density of 2.65 g/cm^3. The case is even worse for concrete lots runoff solids, which would settle 70% slower (assuming a particle density of 1.50 g/cm^3, i.e., the average of Pepple et al., 2011 and Freck and Gilbertson, 1973). However, assuming one particle density for all runoff solids may be oversimplifying. The studies by Gilbertson and Nienaber (1973) and Gilbertson et al. (1975) reported particle density increased with decreases in particle size (Fig. 3). In the Gilbertson and Nienaber (1973) study, the densities reported are for solids that were collected from feedlot runoff; the Gilbertson et al. (1975) data are for solids collected directly from the feedlot surface. In both cases the results were similar – smaller particles (< 45 μm) had densities around 2.3 g/cm^3 but for larger particles density decreased rapidly reaching 1.4 g/cm^3 for particles larger than 2000 μm. This implies that by using one particle density to calculate the settling rate for solids the settling rate of large solids will be overestimated whereas those of small solids will be underestimated. Interestingly, although the particle densities as a function of particle size were similar for both studies (Gilbertson et al., 1975; Gilbertson and Nienaber, 1973) the overall “average” particle densities measured were quite different, which would imply preferential transport of particles during feedlot runoff events.

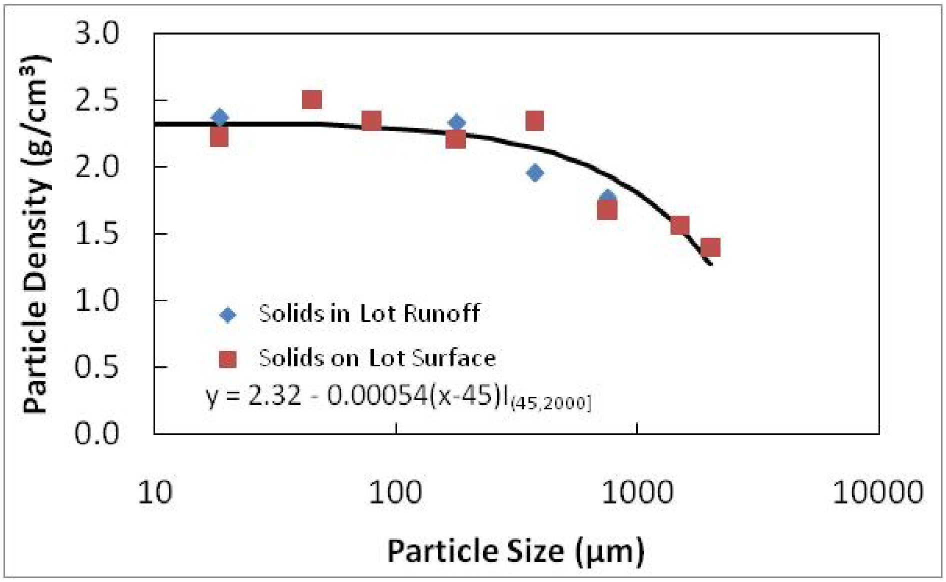

Figure 3. Particle density of solids in lot runoff (from Gilbertson and Nienaber, 1973) and solids on the lot surface (Gilbertson et al., 1975) versus particle density.

This variation in particle size with particle density could also be indicative of differences in nutrient and carbon concentrations in the fractions. Although not investigated in the work of Gilbertson and Nienaber (1973) and Gilbertson et al. (1975), Chang and Rible (1975) presented a study on variation in nitrogen, phosphorus, and crude fiber content with particle size in feces from beef cattle. The COD of each particle size was calculated by assuming all crude fiber was organic carbon and then dividing by a factor of 2.67 based on the ratio in AgNPS (Young et al., 1987). This information is summarized in table 1. In general, crude fiber, and therefore presumably COD, was of greater concentration on larger particles, while nitrogen and phosphorus are more prevalent on smaller particles. Thus, the crude fiber/COD data seems to correspond with the particle density data of Gilbertson and Nienaber (1973) and Gilbertson et al. (1975), i.e., particles with more organic carbon had lower density. Furthermore, this data indicates that sedimentation can be used to effectively reduce the strength, i.e. the chemical oxygen demand, of the wastewater, but that nitrogen and phosphorus reductions need longer settling times due to their association with smaller particles.

Table 1. Variation of crude fiber, nitrogen, and phosphorus content with particle sizes in beef cattle feces (from Chang and Rible, 1975).

Particle Size (mm)	Percent of Manure in Size Fraction	Nitrogen (%)	Phosphorus (%)	Crude Fiber (%)	COD† (%)
> 1.00	30.7	1.7	0.83	43.7	16.3
0.50 – 1.00	9.0	2.2	0.39	58.7	21.9
0.25 – 0.50	6.7	2.5	0.41	32.8	12.2
0.105 – 0.25	6.1	2.7	0.73	27.6	10.2
0.053 – 0.105	3.6	2.8	*	16.6	6.2
< 0.053	43.6	4.9	1.42	10.2	3.8

† COD calculated by assuming crude fiber is all organic carbon and dividing by 2.67 as per Young et al. (1987) in AgNPS

* Value not reported by Chang and Rible (1975)

Particle Size Distribution

As discussed by Gilley et al. (2011) runoff rate has a significant influence on both erosion rate and particle size distribution. This implies that the sediment size distribution will be extremely variable as flow rate is a function of the feedlot runoff hydrograph, which in turn is a function of the storm hyetograph, feedlot characteristics, and the current hydraulic conditions of the feedlot surface. Swanson and Mielke (1973) reiterate this thought, stating that it is extremely difficult to apply empirical formulas to scientifically design solids traps for feedlots because variation in materials available for transport, varying rainfall energies and runoff rates, and changing water temperature and viscosities create large variability in material settleability. Additionally, Møller et al. (2002) found that the relative fraction of large particles decreased and small particles increased with storage time of manure. Typically open feedlots only remove fecal materials from the lot once or twice per year, thus the particle size distribution of the manure would be expected to change with the amount of decomposition that occurred and the rate at which new fecal material is added. Furthermore, as there is a large amount of mixing between the cattle feces and the feedlot soil, particle size distributions from earthen feedlots will be a function of not only the animal waste, but also the soil on which the lot is constructed.

Chang and Rible (1971) performed one of the earliest studies on the particle size distributions of livestock waste. In their study they dry-sieved (for deposited beef manure) or wet sieved (fresh-collected) manure samples to separated particle sizes into six fractions ranging from greater than 1000 μm to less than 53 μm. They found that approximately 56% of solids in manure were sand sized or larger (> 53 μm) and the other 44% were silt and clay sized particles (< 53 μm). Similarly, Gilbertson and Nienaber (1978) used a sieving procedure to study particle size. They found that 60% of solids in cattle manure were silt and clay sized and 40% were sand sized. In a study on the effect feed ration had on beef cattle feces, Frecks and Gilbertson (1974) found that cattle fed a high concentrate ration had a larger percent of solids of silt sized or finer (50%) than cattle fed a high roughage ration (29%). Gilbertson et al. (1975) and Swanson and Mielke (1973) reported particle size distributions, again determined with a sieving procedure, for solids collected from earthen feedlot surfaces. Gilbertson et al. (1975) reported that the material was 30% sand sized or larger, 60% silt sized, and 10% clay sized. Swanson and Mielke (1973) found a particle size distribution of 17% sand sized, 47% silt sized, and 36% clay sized on the lot they monitored. The particle sizes reported by both Gilbertson et al. (1975) and Swanson and Mielke (1973) suggested a substantially smaller sand sized fraction than was found in fresh cattle manure (Chang and Rible, 1971; Gilbertson and Nienaber, 1978). This would tend to follow the suggestion of Møller et al. (2002) that storage time, in this case weathering and decomposition during accumulation and storage on the feedlot surface, may be decreasing the percentage of large particles and increasing the percentage of small particles. Alternatively this could indicate that preferential transport of the smaller particles is occurring (Miner et al., 1966).

Gilbertson and Nienaber (1973), Pepple et al. (2011), and Gilley et al. (2011) all reported particle size distributions for solids transported in feedlot runoff. The Gilbertson and Nienaber (1973) and Pepple et al. (2011) studies reported average particle size distribution from solids transported from multiple runoff events, and in the case of Pepple et al. (2011), from several different feedlots, from naturally occurring runoff events. The Gilley et al. (2011) study used rainfall simulator to generate runoff. Gilbertson and Nienaber (1973) found that 80% of solids were silt sized or finer, while only 20% of all solids were sand sized. Pepple et al. (2011) reported that on average 2% of the transported solids were sand sized, 48% were silt sized, and 50% clay sized. Moreover, Pepple et al. (2011) found no difference in particle size distributions between the solids in the runoff from the concrete and earthen lots they sampled, possibly due to the large event-to-event variation in settling rates. Gilley et al. (2011) reported that the sand sized fractions ranged from 10 to 45% of the particles transported in feedlot runoff while the clay sized fractions ranged from 15 to 55%. Gilley et al. (2011) also reported that median particle size was significantly affected by runoff flow rate, with median particle diameter increasing with increasing runoff rate. This again illustrates that solids transport is a strong function of the hydrology.

Temperature Effects

Gilbertson et al. (1971) and Gilbertson et al. (1980) have stated that solids concentrations in snowmelt feedlot runoff are greater than rainfall runoff. Additionally, feedlot producers in Iowa have noted that more solids tend to escape from the settling basin in winter and early spring than in other seasons. The fact that these issues are seasonal provides an indication that the poorer settleablity and the increased concentrations seen in snowmelt runoff as compared to summer events may be attributed to temperature effects. The fluid density and viscosity are both impacted by the temperature of the runoff; however, in general, the temperature effect on fluid density is relatively negligible, as water density (which is the primary component of feedlot runoff) varies by less than 0.2% from 0°C to 20°C (approximately the range over which runoff temperature could be expected to vary). Viscosity of water on the other hand can vary by almost 44% over the same range, indicating that it has the potential to greatly impact settling rates. This leads to two questions: (1) what effect does temperature (viscosity) have on settling rate, and (2) why are solids concentrations in snowmelt feedlot runoff consistently higher than rainfall runoff?

It is well established that water viscosity increases significantly with decreasing temperature. Similar results were seen by Kumar et al. (1972) for manure slurries. For slurries below 5% total solids Kumar et al. (1972) noted that fluid behavior was Newtonian and viscosities were similar to those of water. Thus for this analysis it was assumed that the viscosity of the runoff was the same as that of water, although differences could occur. The required retention times to settle runoff particles in a one meter deep settling basin were determined. The results indicated that near freezing temperature (0°C) could increase the required settling time by 1.8 times as compared to temperatures near 20°C. This can drastically reduce the performance of the settling basin, especially in removing particles that settle at or near the critical particle settling velocity for which the basin was designed. This indicates that to maintain the same level of performance a longer retention time is required during cold weather than during warm weather. Similarly, the higher viscosity of the runoff may be reducing that amount of “within pen” settling of solid particles by slowing settling velocities sufficiently that solids that would typically settle from the flow before reaching the edge of the pen are now transported to the settling basin. Moreover, this increased viscosity could explain the lava-type flow Gilbertson et al. (1980) reported.

Hindered Settling

Hindered settling is a term used to designate the decrease in fall velocity of sediment in suspension and is believed to be caused by inter-particle forces slowing fall velocity. High sediment concentrations cause significant counterflow of the suspending fluid around the particles, increasing the drag force the settling particles experience, slowing the rate at which particles are settled. Few references have discussed hindered settling in relation to feedlot runoff; however, Pepple et al. (2011) mentioned it as a possible cause for slower settling rates. They noted samples that had high total solids concentrations (particularly samples from concrete lots) often experienced hindered settling (determined based on visual inspection of the settling results). Pepple et al. (2011) only saw hindered settling in one earthen runoff sample. This was a winter snowmelt sample that had a solids concentration of 2.2%. Moreover, the concrete lot runoff samples that experienced hindered settling had solids concentrations exceeding 1.5%. These values are in line with those recommended by Steel (1960) and Sobel (1966) whom suggested that volumetric concentrations of suspended particles exceeding approximately one percent (by volume) experience hindered settling; assuming the particle densities of earthen and concrete lots (discussed above) this equates to solids concentrations of 1.5% and 2.0% (by mass).

Although this research provides a basis for predicting when hindered settling has a significant impact on settling velocities it does little to help model the process. Understanding and modeling the impacts of increased concentrations requires a more detailed description of the mechanisms slowing the settling velocity. Thacker and Lavelle (1977) identified three processes that mechanisms; these were: (1) the retardation that a single particle of sediment experiences due to the counterflow of the suspending fluid, (2) the partitioning of gravitational, drag, and pressure forces between the sediment and suspending fluid, and (3) the modification of the flow field in the vicinity of the sediment particles when other particles are nearby. Moreover, Kumar et al. (1972) found that increases in total solids concentrations positively correlated to increases in apparent viscosity of manure slurry. Utilizing a two-phase flow analysis approach, Thacker and Lavelle (1977) suggested that suggested that the kinematic effects (counterflow of the suspending fluid and the impact of concentration on the pressure field) would reduce settling velocities a factor of $(1\text{-}C)^2$; where C is the volumetric concentration of solids within the fluid. However, they could not extend their analysis to account for dynamic factors. Based on experimental work Maude and Whitmore (1958) suggested that the hindering factor should be $(1\text{-}C)^{\alpha}$ where α is a constant between 4-9 and C is as defined previously.

Here we suggest a value of 9 as Steel (1960) and Sobel (1966) suggested that volumetric solids concentrations of 1% slowed settling velocities by about 10%.

SETTLING VELOCITIES REVISITED

As an alternative to measurement of settling velocities a theoretical approach, Stokes law, could be used to predict settling velocities. Stokes law can be used to calculate the required retention time to settle various sizes of runoff particles. Using solid particles densities of 2.0 and 1.5 g/cm^3 for solids in earthen and concrete lot runoff and assuming that Stokes law applies, the time required to settle sand sized particles (50 μm) and larger is about 0.25 and 0.5 hours for solid particles from earthen and concrete lots respectively (assuming a 1 m settling basin depth). Recalling the particle size distributions reported by the various researchers (Chang and Rible,1971; Swanson and Mielke, 1973; Gilbertson and Nienaber, 1973; Frecks and Gilbertson, 1974; Gilbertson et al., 1975; Gilbertson and Nienaber, 1978; Pepple et al., 2011; and Gilley et al., 2011) we find that a large percentage of particles (47 – 65%) tended to be silt sized (2 – 50 μm). These particles can be settled, but require long retention times to achieve removal. For instance, particles that are 10 μm in diameter require 7 and 13 hours to settle while 3 μm particles require 3 and 6 days. Although not a perfect match, these values are qualitatively similar to the settling velocity distributions reported earlier within this manuscript, especially as this analysis didn’t include the impact hindered settling would have on estimated settling times. These retention times are often too long to be practical at most operations, and as a result these solids tend to settle in containment basins and the additional treatment components used in the runoff control system.

IMPLICATIONS FOR SOLID SETTLING BASIN DESIGN

This brings us to objective three; evaluating the reported settling characteristics of feedlot runoff solids in terms of settling basin design standards and recommendations. This will be done following two different methodologies; the first follows the settling basin design guidelines for open beef feedlots specified by the Iowa Department of Natural Resources (Iowa DNR, 2007) and the second following the recommended design practices recommended in the USDA’s Vegetative Treatment System guidance document on solid-liquid separation (Nienaber et al., 2006).

Iowa administrative code requires feedlot runoff be slowed to a flow velocity of less than 0.15 m/s for a minimum of five minutes, sufficient storage capacity to contain all runoff from a 10-year, 1-hour storm, and one square meter of settling basin surface area for every 2.4 m^3 of runoff per hour. Let’s look at an example using these design criteria; assume a 2-ha (4.95 acre) earthen feedlot near Ames, Iowa (10-year, 1-hour storm of 5.8 cm/hr). Since this is an earthen surfaced feedlot a curve number of around 91 is appropriate for determining the volume of runoff. Based on this value approximately 3.63 cm of runoff should be generated resulting in 726 cubic meters of runoff. Since this is the volume of runoff occurring in one hour a surface area of the settling basin of 300 m^2 is required. Assuming the basin is designed to provide the required five minute detention time and that the basin is at steady flow (inflow = outflow) then the depth of liquid in the basin would be 0.2 m. Since the Iowa administrative code requires a maximum flow velocity of 0.15 m/s at the five minute detention time, a basin length of 45 m is required. Thus in this case the basin dimensions would be 6.6 m wide, 45 m long, and a flow depth of 0.2 m. What is the minimum particle size that would be completely removed in this basin? Based on the depth of flow and hydraulic retention time the basin should capture all particles with a settling velocity greater than 0.00067 m/s. Using Stokes' Law (eq. 1) and assuming a particle density of 2000 kg/m3, a fluid density of 1000 kg/m3, and a viscosity 0.001307 N-s/m2 (~10°C) this corresponds to a particle size of approximately 40 μm. If this had been a concrete lot (curve number of 94 and a particle density of 1,500 kg/m^3) then the design volume would have been 853 m^3, the length would still be 45 m, the required width would be 7.8 m, and the liquid depth would be 0.2 m. In this case the smallest particle size settled would be 57 μm. Thus in both cases the settling basin would be designed to remove approximately all the sand sized particles, but virtually none of the silt.

The USDA NRCS VTS guidance document suggests designing for a solid settling rate of 1.22 m/hr (0.61 m/hr for basins less than 0.61 m deep), for either a 10-year, 1-hour storm event or a 25-year, 24-hour event, a dewatering time of 30-72 hours, and a liquid depth of less than 1 m. Since the basin designed using the Iowa standard had a liquid depth of less than 0.61 m let's assume the more conservative case of a 0.61 m/hr settling rate. In this case it is suggested to design assuming all precipitation will be converted to runoff. Following this suggestion the required surface area would be 1,900 m^2. The storage volume is calculated in two ways, the first is based on the projected runoff volume (1,160 m^3) or from the liquid storage depth (calculated from the settling rate and a user selected detention time, minimum of 0.5 hours) times the basin area (580 m^3). The larger value 1,160 m^3 is selected. Again assume the basin is at steady flow conditions, the liquid depth would be the design volume divided by the surface area 0.6 m and the detention time would be one hour (basin volume divided by flow rate). Following the above procedures the minimum particle sizes that would be projected to be removed would be 20 μm for an earthen lot and 28 μm for a concrete lot, or roughly half of all silt sized particles.

These analysis have several cavettos; the first being that the settling process was well modeled as discrete particle settling, the second that basins were modeled as being at steady flow conditions, and the third was the settling was ideal, i.e., there were no turbulence effects that would cause particle resuspension or delay settling, and the forth is the viscosity was assumed identical to that of water. Redoing the problems above and assuming approximately 0°C (still assuming viscosity is the same as water) the critical particle diameters would be 46 μm and 65 μm for earthen and concrete lot settling basins designed to meet Iowa criteria and 23 μm and 32 μm for the USDA VTS guidance document method.

Are these removals sufficient? This all depends on what the particle size distribution in the feedlot runoff is. One possible way of evaluating if this level of performance is acceptable is by looking at the particle size distribution data of Gilley et al. (2011) and evaluating what solids concentration reductions would be expected. As can be seen, this value would range from approximately 15-85% for settling basins designed according to the minimum Iowa DNR requirements or 20-90% for settling basins designed according to the USDA VTS recommendations, depending on the flow rate of the runoff. Referring back to our example of an earthen lot (curve number 91) lot runoff rates of 91 L/min-m would be expected. This value is substantially greater similar than the runoff rates monitored by Gilley et al. (2011), but assuming there highest flow rate (20 L/min-m) gives solids representative of those present on the lot, than 80-90% of all solids transported could reasonably be expected to settle; however, less intense storms would result in lower runoff rates. One hour storms with intensities of 0.75 and 2.54 cm/hr would produce a runoff rates varying from 0.5 to 21.5 L/min-m. These low flow runoff events would result in much greater variation in settling basin performance (when evaluated as percent reduction in solids concentration), and since 90% of all storms near Ames are less than 2.54 cm in total size could play a major role in the monitored performance of the basin. Additionally, it would also help explain why the samples of Pepple et al. (2010) displayed substantially poorer settling characteristics than the solids obtained directly from the feedlot in the Lott et al. (1994) study and the electronically sampled runoff in the Gilbertson and Nienaber (1973) study, i.e., it’s probably that the samples of Pepple et al. were more similar to the 0.67 L/min-m flow rate, those of Lott et al. (1994) were similar to the 20.4 L/min-m flow rate, and those of Gilbertson and Nienaber (1973) were in the 6-13 L/min-m range.

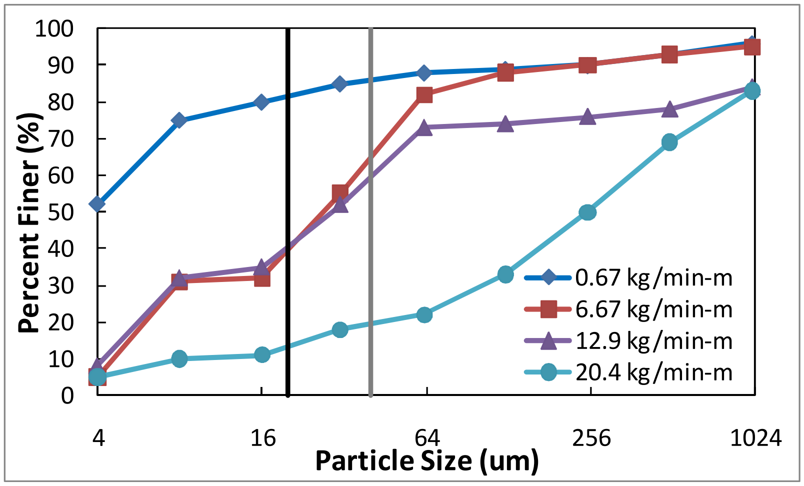

Figure 4. Particle size distributions of solids in runoff from an earthen beef feedlot near Clay Center, Nebraska at four runoff differing runoff rates per width of plot. The black line represents the critical particle diameter for earthen lot settling basin designed using the Data from Gilley et al. (2011).

CONCLUSIONS

Primary treatment of feedlot runoff typically relies on solids removal, usually through sedimentation techniques. Rigorous engineering design of sedimentation basins must consider flows into, within, and out of the settling structure, and the settling characteristics of the solids in the runoff. In this review typical settling properties of solids in runoff from concrete and earthen lots was reviewed. The results show that solids in runoff from concrete lot settle substantially slower than from earthen lot runoff. This was in large part caused by differences in average particle density between the solids in the runoff (1,500 kg/m^3 for concrete runoff, 2,000 kg/m^3 for earthen lot runoff). The large differences in average particle density can be explained by the amount of soil mixing with cattle feces and being eroded from the earthen surface. Particles size distributions exhibited substantial variation, both between and within lots, which appear to be attributable to event hydrology. More work relating particle size distributions to runoff rates are required to verify this conclusion. Current settling basin design standards are then evaluated in light of the discussed settling characteristics; overall results indicated that the USDA NRCS settling basin design recommendations would settle particles of 23 μm and 32 μm and larger from earthen and concrete lots respectively; whether this treatment is adequate depends on the additional treatment the runoff receives. The results also indicate that evaluating settling basin performance as percent reductions in total solids my lead to a large variation in reported performance as it is very dependent on the particle size distribution of sediment in the inflow.

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Chapter 4. Development of a Runoff and Sediment Routing Model for Open Lot Beef Feeding Facilities

Abstract. *Feedlot runoff is a potential environmental contaminant and requires proper management to minimize impact on water quality. In designing runoff management systems accurately assessing the amount of runoff that will be generated is of premier importance. Along with overall quantity of runoff, the temporal pattern, both throughout the year and within the storm event, can have large implications for sizing control system components, in determining the performance the control system achieves, and in the overall pollution potential of the feedlot. This review summarizes the hydraulic properties of the feedlot surface, specifically focusing on variables that impact the total volume of effluent generated and the resulting amount of sediment transported. The work cumulates in development of a feedlot runoff routing model with a sediment transport/erosion component. Results are compared to monitored solids concentrations from a feedlot near Ames, Iowa to validate model performance. The feedlot runoff and sediment routing is used to assess the impact of various feedlot design characteristics, including feedlot area, aspect ratio, and slope, on solids transport from the feedlot surface. This model can be used to evaluate the risk that feedlot runoff poses to water quality, for prioritizing feedlots that are in need of enhanced runoff control systems, and to evaluate the hydraulic and sediment loadings that a runoff control system is required to handle.*

Keywords. *Feedlot runoff, feedlot hydrology, cattle manure, solids transport, erosion, modeling*

INTRODUCTION

Concern over water pollution associated with animal waste has increased with the intensification of livestock production. The passage of the Federal Water Pollution Control Act Amendments in 1972 placed the U.S. Environmental Protection Agency (EPA) in charge of developing runoff control guidelines (Anschutz et al., 1979). As a result, the EPA released the Effluent Limitation Guidelines (ELGs), which described the design and operating criteria for concentrated animal feeding operation (CAFO) waste treatment systems (Sweeten et al., 2003). Designing waste management systems that meet these guidelines while minimizing construction costs requires accurate estimation of the amount of waste generated. Along with understanding the hydraulic constraints placed on the waste management system it is also necessary to estimate the nutrient and solids loadings the system's treatment components will encounter. As shown in Andersen et al. (2011), many nutrient concentrations can be estimated based on knowledge of total solids content, thus physically based process models that link erosion to feedlot hydrology could be used to estimate nutrient losses from the feedlot.

A feedlot is subject to the same erosion producing rainfall as the adjacent land, and although conditions of the feedlot and the surrounding surface may differ drastically, the effects of rainfall on solids transport and the erosion process are similar (Swanson et al., 1971). On an average annual basis this erosion is a function of slope angle and length, infiltration rate, and physical properties of the soil (Zing, 1940). However, the intensity, amount, and duration of rainfall can have a profound effect on the rate of the resultant runoff, and therefore erosion (Ayers, 1936). Thus, the objective of this work was to develop a model of the feedlot surface (using inputs of feedlot surface type (earthen or concrete), average slope, size, aspect ratio, and a precipitation hyetograph) that is capable of predicting runoff volumes and sediment mass transport from the feedlot that can be used to aid in the design of solid settling basins and runoff control systems. Specifically, model development will (1) discuss the hydraulic properties of the feedlot surface, (2) discuss the various methodologies that have been used in modeling runoff volumes, (3) propose a methodology for constructing a hydrograph from the feedlot surfaces, (4) develop a relationship between sediment transport and runoff flow rate, and (5) evaluate the implications this model has for laying out a feedlot to minimize sediment transport.

FEEDLOT HYDRAULIC PROPERTIES

Properties of the Feedlot Surface

The physical properties of the soil and manure pack (thickness, bulk density, water holding capacity, moisture content, hydraulic conductivity, infiltration rate, etc.) determines the water balance in the feedlot area and is responsible for partitioning precipitation into storage, runoff, and leaching volume. Mielke et al. (1974) suggested that three layers develop in the soil profile in a feedlot; a layer of manure accumulation, a black interface layer of mixed organic and mineral soil, and the native top soil. Moreover, he suggested that this interface layer was primarily responsible for limiting hydraulic conductivity. This layer forms through physical, chemical, and biological processes such as compaction from hoof traffic, plugging of soil pores, dispersion of clays from the high sodium and potassium levels, and biofilm development (Mielke et al., 1974, Schuman and McCall, 1975, Miller et al., 1985, Rowsell et al., 1985, Barrington et al., 1987; McConkey et al, 1990). This description of the feedlot soil profile has generally been accepted (Miller et al., 2008; Olson et al., 2006; Maule and Chi, 2006), although more recent work by Cole et al. (2009) divided the manure accumulation into two layers, an upper dryer layer and lower wetter layer, although they propose that this division may be weather dependent with the boundary changing due to environmental conditions. Underneath this manure accumulation layer Cole et al. (2009) found a black interface layer that would limit seepage. In either case (i.e., the profile of Mielke et al., 1974 or that of Cole et al., 2009), the manure-soil hydrologic response expected would be similar; the upper layers (manure) should act as a sponge soaking up added moisture and the compacted soil-manure interface as an impermeable, or very slowly permeable, layer (Mielke et al., 1974). This is not to say leaching from a feedlot surface does not or cannot occur, but rather that on the time scale of the precipitation event seepage through this interfacial layer should be negligible in the overall water balance. For instance, Mielke and Mazurak (1976) reported feedlot infiltration rates of 0.12 cm/day while values from McCullough et al. (2001) ranged from 0.05 to 0.16 cm/day. This is also true of concrete lots as no infiltration could occur; although in both cases significant fractions of precipitation could be stored in the accumulated manure depending on its moisture holding characteristics and its current moisture content.

Another unique property of the feedlot surface is that in addition to precipitation, it also receives moisture through cattle defecation. The ASABE manure characteristics standard (ASABE, 2005) can be used to provide an estimate of the average annual addition of water to the feedlot surface. This is a function of animal stocking density and is presented as such in Fig. 1 (assumes two cattle feed out cycles per year). As can be seen, the moisture addition in feces and urine can be quite large, even at 25 m^2/head (typical stocking density for earthen lots in Iowa) approximately 40 cm/year of water are added to the feedlot surface; for concrete lots, which often stock at densities of around 12 m^2/head up to 82 cm/yr can be added. This amount of added moisture is important to consider when evaluating feedlot surface properties as it can increase moisture levels; for instance in Iowa annual precipitation ranges from 63 to 102 cm (25 to 40 inches) thus moisture from animal defecation can account for between 30 and 60% of the average annual moisture the feedlot surface receives.

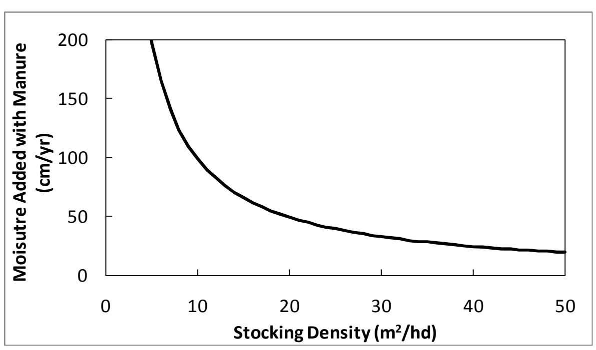


Figure 1. Moisture additions to the feedlot surface resulting from cattle defecation. Calculated based on ASABE Standard 384.2 assuming two cattle grow outs per year

Predicting Runoff Volumes

Researchers (Gilbertson, 1980; Clark et al., 1975b) have suggested that the Soil Conservation Service (SCS) Curve Number (CN) method and linear regression equations are viable methods of predicting runoff volumes from beef feedlots. Clark et al. (1975a) utilized the regression method to show rainfall-runoff relationships from six feedlot locations. Based on these equations Gilbertson et al. (1980) stated that between 0.75 and 1.5 cm of rainfall will be retained on the feedlot surface and between 36 and 86% of any additional rainfall will runoff, with values fluctuating due to lot antecedent moisture conditions, feedlot shape, slope, and the type of feedlot surface. As an alternative method, numerous researchers have utilized the curve number method. Vanderholm and Dickey (1980) recommended values of 95 to 99.9 for concrete paved dairy lots and a value of 90 for paved beef cattle lots (Dickey and Vanderholm, 1977), suggesting that the greater manure accumulation on the surface of the beef lot resulted in greater retention of precipitation. Work from Gilley et al. (2011) suggested similar results, finding that for wet earthen feedlots a curve number of 90 was appropriate and that greater accumulation of unconsolidated surface materials reduced runoff volumes. Miller et al. (2003) studied runoff from unpaved lots near Alberta, Canada finding that curve numbers varied from 52 to 96 for runoff events, mostly due to different amounts of water storage within the feedlot manure pack, which they propose acted like a sponge, absorbing the initial rainfall until it became saturated. This follows the suggestion of Clark et al. (1975b) that the percentage of rainfall that runs off is proportional to the moisture deficit (evaporation minus precipitation) of the region. Similarly many researchers have found that feedlot curve numbers can vary substantially (Table 1) with different weather and storm patterns. This analysis is supplemented with Fig. 1, which used the precipitation and runoff data of Swanson et al. (1971), Swanson and Mielke (1973), Miller et al. (2004), Andersen et al. (2012), and Kreis et al. (1972) to determine which curve number and linear equation best fit the relationship between storm size and runoff depth. Both the curve number and linear equation fit the data similarly, explaining 73% of the variation in runoff depth. The ideal curve number was determined to be 91 and the linear regression equation suggested that 1.2 cm of precipitation was required to initiate runoff at which point 74% of all additional precipitation became runoff. However, there was again substantial variability about these relationships. This has led researchers to question the use of a standardized curve number for modeling feedlot runoff, and instead investigate the use of antecedent rain indexes and water balances on the manure pack to estimate runoff and speculate about the use models to simulate the dynamic process of infiltration and runoff.

Table 1. Runoff curve numbers reported in literature for describing the volume of feedlot runoff from sites with varying stocking densities, feedlots surfaces, and weather conditions.

Author	Feedlot Conditions	Location	Curve Number	% Variation
Kennedy et al. (1999)	Unpaved, 17 m^2/head, Rainfall	Alberta	55 - 83	51
Kizil and Lindley (2002)	Pond Ash, 46 m^2/head, Rainfall	North Dakota	82 - 97	18
Swanson et al. (1971)	Unpaved, Rainfall Simulator	Nebraska	76 - 98	29
Swanson and Mielke (1973)	Unpaved, Rainfall	Nebraska	73 - 100	37
Miller et al. (2004)	Unpaved, 18 m^2/head, Rainfall	Alberta	59 - 95	61
Andersen et al. (2012) – CN IA 1	Unpaved, 30 m^2/head, Rainfall	Iowa	77 - 100	30
Andersen et al. (2012) – CN IA 2	Unpaved, 16 m^2/head, Rainfall	Iowa	77 - 98	27
Andersen et al. (2012) – NW IA 1	Unpaved, 21 m^2/head, Rainfall	Iowa	94 - 100	6
Andersen et al. (2012) – NW IA 2	Paved, 7 m^2/head, Rainfall	Iowa	73 - 100	37
Kreis et al. (1972)	Soft chalky bedrock, 11 m^2/head, Rainfall	Texas	79 - 99	25

One such water balance model is that of Maule and Chi (2006), and although their model met with only limited success, it provides a framework for physically based feedlot runoff models. Their model used a moisture balance to calculate the retention factor used in the SCS CN method. The basis of the water balance was that water inputs from the both precipitation and cattle defecation and losses to evaporation change the moisture content of the manure pack, and that the available water holding capacity of the manure pack is equal to the SCS retention factor. In developing this moisture balance Maule and Chi (2006) assumed that seepage from the manure pack was negligible. Although the water balance method showed promise and was more successful than a either a constant curve number or a curve number based on an antecedent precipitation index (Maule and Chi, 2006), more information on the hydraulic properties (conductivity, water retention, evaporative drying characteristics, porosity, wetting suction, rewetting characteristics) of the manure pack are needed, limiting implementation of this methodology.

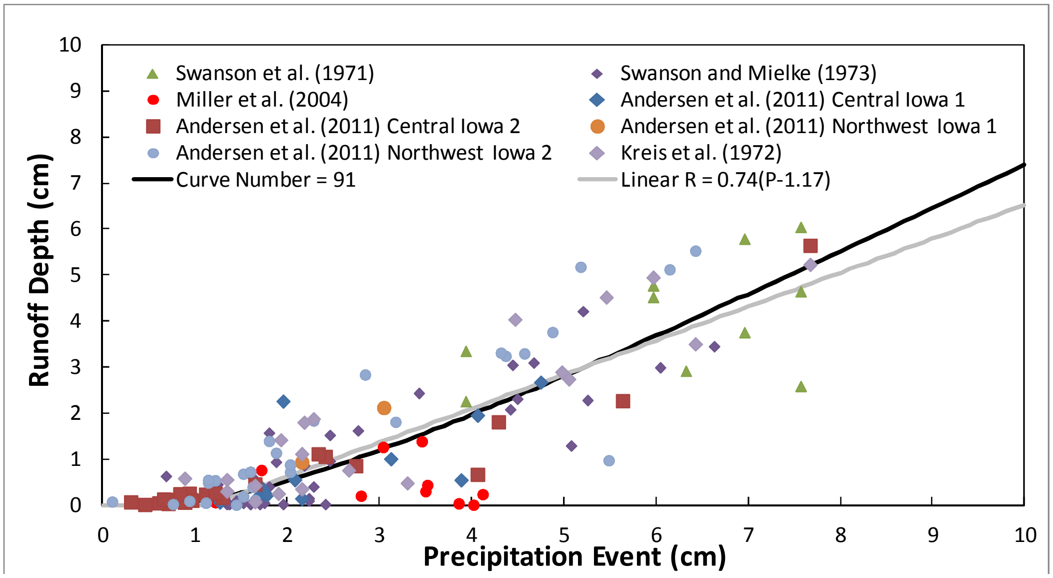


Figure 1. Monitored runoff depth versus precipitation event size. Data from Swanson et al. (1971), Swanson and Mielke (1973), Miller et al. (2004). Andersen et al. (2011), and Kreis et al. (1972). The SCS curve number and a linear regression equation were fit the observed data. Model fitting suggested that the best curve number to use was 91 and the linear relationship indicated that 1.17 cm of precipitation were required to initiate runoff and thereafter 74% of all additional precipitation was converted to runoff; both equation had R^2 values of 0.79.

Modeling the Runoff Hydrograph

Along with knowing the amount of runoff that occurs, proper analysis of settling basin performance and solids transport from the feedlot requires information on the runoff hydrograph (Lott et al., 1990). Little research has focused on this area; however, work by Swanson et al. (1971) and by Gilley et al. (2011) have shown that erosion from a feedlot surface is proportional to the flow rate of runoff across the surface, i.e., that the transport of sediment from the feedlot surface is in general transport limited. Moreover, Lott et al. (1990) suggested a similar idea, stating that experience in Australia has shown settling basin weirs are more prone to clogging after intense rainfall events, possibly due to increased momentum carrying more manure into and through the settling basin. The link between solids transport and flowrate reported by these researches seems plausible from a mechanistic standpoint as the feedlot surface is often covered with highly erodible particles; however, utilizing a relationship between flowrate and solids transport to predict feedlot runoff solids content requires a flow routing method be used to generate the runoff hydrographs.

Several methods have been proposed to generate runoff hydrographs; these include hydrograph fitting, kinematic flow routing, and SCS synthetic hydrograph generation. Using hydrograph fitting would require the generation of large datasets in which both the precipitation hyetograph and the runoff hydrograph are monitored prior to interception by the runoff control system. Although example hydrographs from earthen and concrete lots have been reported in literature (Miner et al., 1966) insufficient information is provided to construct a unit hydrograph based on their findings and to generalize it to feedlots of differing size and slope. A second approach, using kinematic wave theory was proposed by Lott et al. (1990). Although many of the underlying assumptions of kinematic wave theory are plausible for feedlots (pen surface is relatively uniform with no significant irregularities, precipitation hyetograph across the feedlot surface would be similar, in most cases backwater effects would be negligible upstream of the sedimentation system), an accurate Mannings coefficient is required (Lott et al., 1990). The value of Mannings coefficient is unknown and probably varies with different pen surface conditions. Thus, at this time the SCS synthetic unit hydrograph approach as outlined by Haan et al. (1994) seems appropriate.

In the SCS synthetic unit hydrograph approach the first step is to estimate the time to peak of the hydrograph. This can be estimated using the SCS Method (1975) as shown in Eq. 1. In this equation T_p is the time-to-peak of the hydrograph in minutes, Δt is the duration of the unit excess rainfall in minutes, L is the length of the longest flow path in meters, CN is the runoff curve number (which could be adjusted based on the available water holding capacity of the feedlot surface), and *slope* is the average slope of the feedlot in m/m. This value can then be used in Eq. 2 to calculate the peak flow rate. In this equation q_p is the peak flow rate in cubic meters per second per centimeter of effective precipitation, A is the area of the feedlot in square meters, and T_p is the time of concentration in minutes. The SCS Dimensionless Unit Hydrograph can then be used to generate a unit hydrograph specific to the feedlot. Eq. 3 provides a normalized equation which can be used to approximate the SCS hydrograph at different points in time, in this equation U is the flow rate of the unit hydrograph, in m^3/s per cm effective precipitation, q_p is the peak flow rate (m^3/s-cm effective precipitation) as calculated in equation 2, t is the time in minutes, and t_p is the peak time (minutes) as calculated in equation 1. To facilitate programming, the time to peak should be adjusted to occur at the closest multiple of the time-step used in the model and total flow for the unit hydrograph should be adjusted to ensure that it is equal to the equivalent of 1 cm of runoff from the contributing drainage area. Total flow is checked using Eq. 4, where Q should be 1 cm, Δt is the time-step used in the model in minutes (chosen as 5 minutes here), U_i is the flow rate of the unit hydrograph at each point in cubic meters per second per cm, A is the area of the feedlot in square meters, and 6000 is a conversion from meters to centimeters and minutes to seconds. If this equation is not within the desired tolerance (0.0001) the peak flow is adjusted and then tolerance rechecked. This process should be iterated until the tolerance criterion is satisfied.

$$T_p = \frac{\Delta t}{2} + \left[\frac{L^{0.8}\left(\frac{1000}{CN} - 9\right)^{0.7}}{73.45\sqrt{slope}} \right] \tag{1}$$

$$q_p = \frac{A}{8000 T_p} \tag{2}$$

$$U = q_p \left(\frac{t}{t_p} \exp\left(1 - \frac{t}{t_p}\right) \right)^{3.822} \tag{3}$$

$$Q = \frac{6000 \Delta t \sum_{i=1}^{n} U_i}{A} \tag{4}$$

To use the unit hydrograph approach, estimates of effective precipitation for each time step are required. This can be generated by using a storm hyetograph and the SCS curve number method (or a feedlot surface water balance method) to estimate the amount of precipitation during each time step that would be converted to runoff. Using the SCS curve number method this can be accomplished by calculating cumulative precipitation and using the curve number method to determine cumulative effective precipitation at a given time step. The amount of effective precipitation for the current time step is then calculated by subtracting off the cumulative effective precipitation of the previous time step. The water balance method would be performed similarly, although in this case there would be no runoff until the available soil storage capacity was exceeded, at which point all additional rainfall would be considered effective precipitation. The runoff hydrograph is then generated by convolution of the excess rainfall hyetograph and the unit hydrograph. This is done using Eq. 5. In this equation q_n is the flow rate of the n^{th} time increment of the runoff hydrograph in cubic meters per second, P_m is the effective precipitation, in cm, occurring during the m^{th} time increment, and U_{n-m+1} is the value of the *n-m+1* time increment of the unit hydrograph, *M* is the number of increments that have excess rainfall, *n* is the time increment flow is being calculated for, and *m* is a count variable that is used to sum all effective precipitation increments that effect the flow of the current time interval.

$$q_n = \sum_{m=1}^{n \leq M} P_m U_{n-m+1} \tag{5}$$

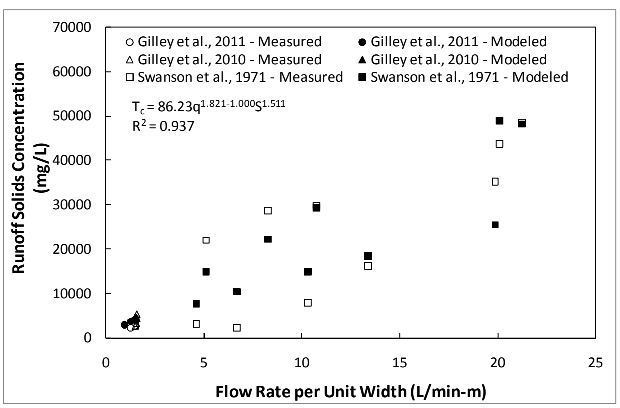
Estimation of Solids Transport

Several theories have been presented on erosion, but the prevailing sentiment among process-based erosion models is that sediment transport capacity is the fundamental concept in determining detachment and deposition processes. Building of this conceptual framework began with the work of Ellison (1944, 1947a, 1947 b, and 1947 c) who proposed dividing erosion into four sub-processes, (1) detachment by raindrop impact, (2) transport by rain splash, (3) detachment by surface flow, and (4) transport by surface flow. Meyer and Wischmeier (1969) proposed that sediment transport was either detachment or transport capacity limited. Since then the concept of limited sediment transport capacity of overland flow has been extensively applied in many physically based soil erosion models (Foster and Meyer, 1975; Beasley et al., 1980; Foster et al., 1995). Prosser and Rustomji (2000) focus on a simple transport capacity model, given as Eq. 6, which has been widely applied to hillslopes. In this equation q_s is the sediment transport capacity (g/min-meter width), q is the flowrate (L/min-meter width), S is the energy gradient (approximated as the surface gradient in %), and k, β, and γ are empirically derived constants. This equation was modified slightly for this analysis; both sides of the equation were divided by flow rate per unit width to solve the equation for runoff solids concentration (Eq. 7). In this case the concentration is in mg/L and k handles the appropriate unit conversions. For this approach to be successful it is required that solids transport is flow capacity limited, that is, there is sufficient erodible particles available on the feedlot surface to satisfy the transport capacity. According to the work of Gilley et al. (2011) this assumptions is reasonable, as feedlot surfaces typically have a large supply of highly erodible material available for transport.

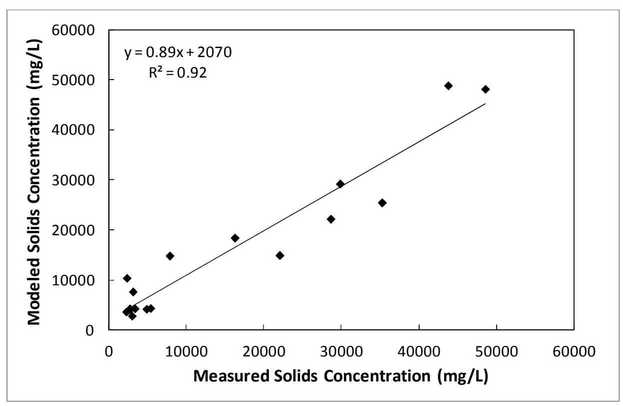
$$q_s = kq^{\beta} S^{\gamma} \tag{6}$$

$$C = kq^{\beta - 1} S^{\gamma} \tag{7}$$

The erosion model (Eq. 7) was calibrated for feedlot sediment concentrations using data from Gilley et al. (2010), Gilley et al. (2011), and Swanson et al. (1971). These authors reported sediment transport and runoff from simulated rainfall events on feedlots of various slopes ranging from 4.8 to 13% and flow rates ranges from just above 0 to about 25 L per minute per meter plot width. Average solids concentrations were calculated by dividing cumulative sediment transport by total runoff. The fitted equation is noted in Fig. 5a; also shown (Fig. 5b) is a plot of measured versus modeled concentrations. In general, this equation showed a reasonable ability to fit the measured data describing more than 93% of the total variability of the solids concentration in runoff from the feedlot. In the plot of measured versus monitored concentration data the best fit line's intercept was not significantly different than 0 and the slope of the line was not significantly different than 1 (α = 0.05) indicating the model performance was adequate. Moreover, the calibrated coefficients β and γ are within the ranges recommended by Prosser and Rustomji (2000) and near their final recommendation of 1.5 for β and γ (we found β = 1.821 and γ = 1.511 respectively); however, there are several limitations to this model. Namely, the model assumes sheet flow over the feedlot surface, although in some cases this may be true, especially on the smaller plots used in generating these data sets, on actual feedlots runoff might be more prone to channeling changing the relationship between relationship between solids concentration and flow rate. For instance, Miner et al. (1966) suggested that under channeling flow conditions runoff could be less polluted due to less interaction between the soil and the runoff water.



(a)



(b)

Figure 5. (a) Solids transport capacity as a function of runoff flow rate and feedlot slope. Open symbols represent measured data and filled symbols represent modeled data using the calibrated equation. Data used is from Gilley et al. (2011), Gilley et al. (2008), and Swanson et al. (1971). (b) Comparison of measured versus modeled solids concentrations shown with the best-fit line. The intercept was not significantly different than 0 and the slope was not significantly different than 1 (α = 0.05) indicating adequate model fit.

Although these controlled plot studies provide detailed information on the relationship flow rate and total solids, they do little to illuminate how these solids were partitioned between suspended and dissolved solids. As illustrated above, solids concentrations are expected to increase with greater flow rates and steeper slopes, this is thought to be primarily due to increased suspended solid transport as steeper slopes and greater flow rates result in greater fluid velocities, larger shear forces on the soil surface, and greater turbulence to mix the sediment into the flow. However, the opposite trend may be expected for dissolved solids, that is increased velocities may lead to decreased concentrations due to less contact time between the flowing water and the feedlot surface (Miner et al., 1966). In addition to the impact of reduced contact time, larger storm events are often cited as diluting dissolved solids content in the runoff (Malouf, 1970). To test the impact of dilution on dissolved solids concentration we regressed the percent of the total moisture the feedlot surface received due to cattle defecation (cattle defecation moisture divided by annual precipitation plus cattle defecation moisture) against average total dissolved solids concentration in the runoff. This correlation was tested using the data from the six sites presented in Andersen et al. (2009) and those reported by Lorimor et al. (2003), Yang and Lorimor (2000), Edwards et al. (1986), Woodbury et al. (2002), and Kreis et al. (1972). The amount of moisture added by cattle defecation was calculated based on the ASABE manure characteristics standard (ASABE, 2005) assuming two feed outs per year, except for the Edwards et al. site where the author reported that only one feed out had occurred. Results indicated that percent moisture added from cattle defecation and average dissolved solids concentrations in the runoff were significantly correlated ($r = 0.8888$, $p < 0.0001$) and that this correlation was quite strong (figure 5) as evidenced by the 201 mg/L (Standard Error ± 34) increase in dissolved solids concentration for every one percent increase cattle defecation moisture had on the moisture balance of the lot surface. This slope of this regression line was significantly different than zero ($p = 0.0003$); however the intercept (-1703 mg/L) was not ($p = 0.2342$).

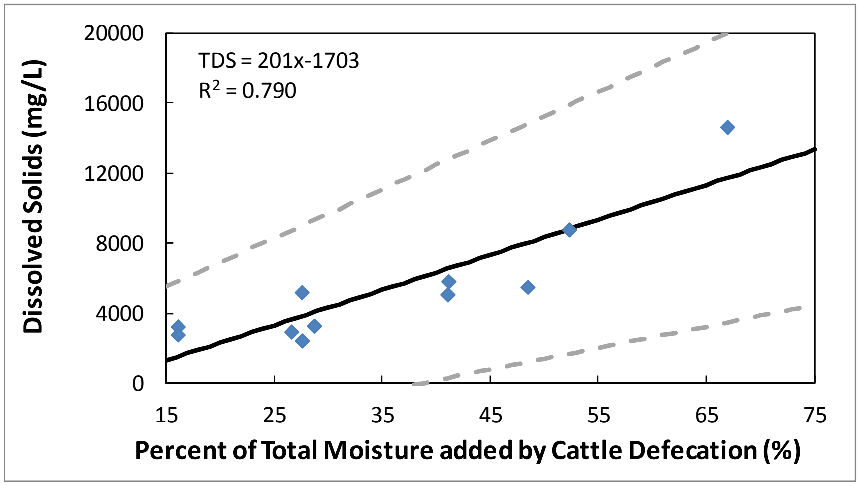


Figure 6. Regression between the average dissolved solids content of feedlot runoff and the percent of the moisture the lot receives from cattle defecation (moisture addition from defecation divided by annual rainfall plus moisture from cattle defecation). Moisture from cattle defecation calculated using ASABE manure characteristics standard. Dashed lines represent 95% confidence intervals.

Model Validation

The runoff and erosion model described above was implemented in java and added to the ISU-VTA model to perform validation testing. Unfortunately, no data sets where runoff rates and sediment concentrations from production feedlots were available to validate this model; however, average annual solids concentrations in runoff from feedlots have been reported on numerous lots (Lorimor et al., 1995; Andersen et al., 2012; Kreis et al., 1972; and Gilbertson and Nienaber, 1973). These data sets provide average solids concentrations for feedlot runoff from lots of varying sizes, slopes, and shapes under different climatic conditions. At each feedlot location (five Iowa locations, a Nebraska location, and a Texas location) we utilized our hydrology-erosion model to predict average total solids concentrations in feedlot runoff over a ten year period. The hydrology-erosion model was input with site specific data (feedlot size, aspect ratio, and slope) based on the author's description of the feedlot. Climatic data (precipitation and daily high and low temperatures) were obtained for each location for the period of 2000-2009 utilizing online sources (http://mesonet.agron.iastate.edu and http://www.wunderground.com) reported for nearby locations. The average total solids concentration for each event was calculated as the runoff event's flow-weighted average solids concentration; runoff events smaller than 10 m^3 were discarded as none of the references reported solids concentrations in samples from events smaller than this magnitude. The minimum solids concentration for each event was set as the dissolved solids concentration calculated from the regression equation listed in Fig. 6. The arithmetic average total solids concentration was then calculated and compared to the average solids concentration reported in feedlot runoff from each site (Fig. 7).

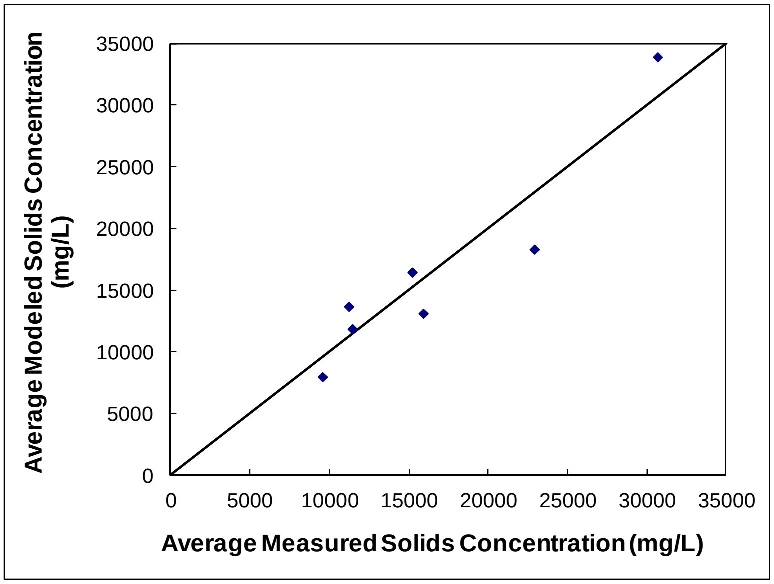


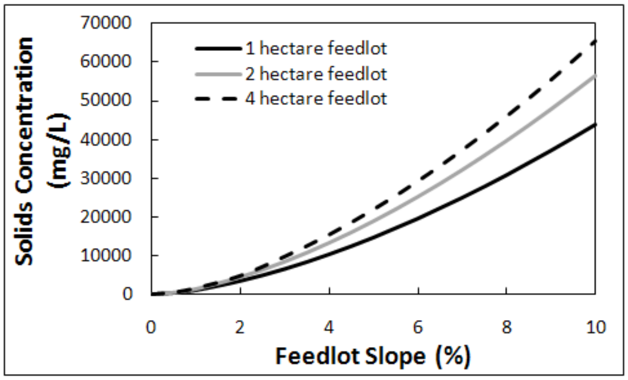
Figure 7. Comparison of average annual measured and modeled total solids concentration in feedlot runoff from seven production feedlots. Data from Andersen et al. (2012), Lorimor et al. (1995), Gilbertson and Nienaber (1973), and Kreis et al. (1972) The line shown represents the 1-to-1 line.

Model performance was evaluated by regressing the average modeled solids concentration against those measured at the site. The resulting regression line had a slope of 1.035 which was not significantly different than zero ($\alpha = 0.05$) and the intercept (-783) was not significantly different than 0 ($\alpha = 0.05$). These results indicate that the model does not show a bias in predicting solids concentrations and appears to be performing well; however, at several of the sites modeled average concentrations differed from measured concentrations by up 25%. Moreover, the slope and intercept of the regression line exhibited substantial uncertainty as 95% confidence intervals for slope were 0.603 and 1.467 while those of the intercept were -8,600 to 7,000. Despite this uncertainty, in general it appears that the model is providing a reasonable prediction solids transport in feedlot runoff and as such may provide useful information on the impact on how different feedlot layouts affect solids transport and pollution potential.

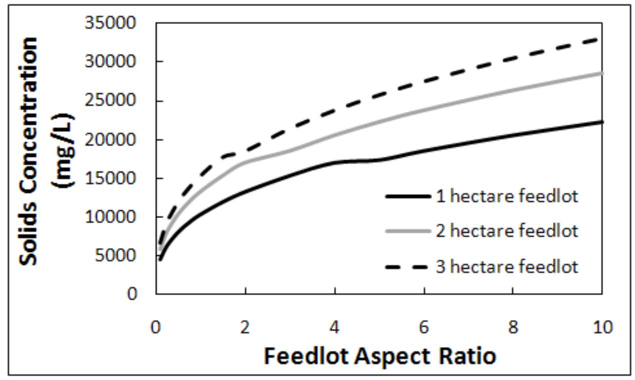
IMPLICATIONS FOR MANAGING THE LOT SURFACE

This leads to the question, given this information how should we design and manage the lot to minimize solids transport? Based on figure 5 it is clear that the flow rate of effluent across the feedlot surface should be minimized by diverting clean water around the feedlot. This need is further emphasized when one considers that the soil detachment rate is usually considered to be proportional to the difference between the sediment transport capacity and sediment load in the flow, and since the outside runoff water would be relatively clean the erosion rate from the feedlot surface would be high. Along with this consideration other measures that reduce flow rates could also be utilized. Based on equations 6 and 7 these measures would include minimizing the size of the feedlot to limit extra runoff from the contributing drainage areas, i.e., stocking cattle at the recommended density, minimizing the slope of the lot to slow the flow rate (although sufficient slope to encourage uniform drainage and maintain a well drained feedlot need to be maintained), and adjusting feedlot shape to minimize the length-width ratio of the feedlot (shorter slope length and less contributing drainage area), or adding settling basins within the feedlot to break up longer slop lengths.

To illustrate these concepts and to better understand effects of the various design variables the developed model was utilized. The first variables investigated were feedlot size and slope. This was done by varying these two parameters while holding storm size and intensity constant. Results (Figure 6a) of this investigation are presented as total solids transported for a 2.54-cm, 1-hour, uniform intensity storm. A feedlot curve number of 91 and a feedlot aspect (length-to-width) ratio of one were used. Results showed that solids concentration increased exponentially with slope, so minimizing the feedlot slope is critical. Lot size increases also increased solids concentration due to greater upslope contributing area, but in this case increases were logarithmic as doubling lot size did not double contributing flow length. Figure 6b supplements this analysis by analyzing the impact of the feedlot aspect ratio (Length to width ratio). This analysis is also presented for a 2.54-cm, 1-hour storm, a feedlot curve number of 91, and a lot slope of 4%. This figure shows the logarithmic increases caused by increasing the drainage length. The slight discontinuities in this graph are caused by incremental change in the peak hydrograph time, which was required to be an increment of the five minute time step used in the analysis.



(a)



(b)

Figure 6. (a) Effects of feedlot size and slope on the solids transport from the feedlot surface on a per hectare basis. (b) Effects of feedlot aspect ratio (length-to-width) and feedlot size on the solids transport from the feedlot surface.

In addition to these designer controlled properties, uncontrollable hydraulic properties also play a key role. To illustrate this effect we calculated the estimated erosion from various intensity storms. In all cases storms were modeled to last for an hour, thus each storm event was of a different magnitude. To make results comparable, flow weighted average solids concentrations are presented (Figure 7). In this case the feedlot slope was specified as 4%, the aspect ratio at 1 and the runoff curve number as 91. This plot illustrates that the larger flow rates produced by more intense rainfall events increases the transport capacity of the flow and with it projected erosion.

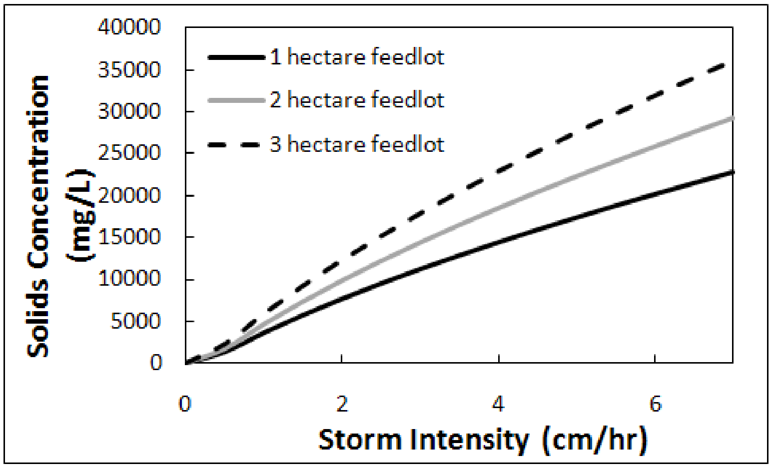


Figure 7. Total solids concentrations in feedlot runoff from a 1-, 2-, and 3-hectare feedlot as a function of storm intensity for a 1-hour, uniform intensity storm. Results are for a feedlot with slope of 4%, an aspect ratio of 1, and a curve number of 91.

CONCLUSIONS

Design of open lot runoff control systems to meet environmental guidelines while minimizing construction and operation costs requires accurate estimation of the runoff volumes. When containment basins are used the estimated volumes most relevant are at the time scale of the application schedule; however, other treatment options, such as sediment basins and vegetative treatment systems, respond to runoff at an event-by-event basis increasing the importance of accurately estimating runoff from individual events. Available literature on estimating feedlot runoff volumes was reviewed; the results indicated that while a curve number of 91 seemed appropriate, substantial variation about this value existed with reported values ranging from 55 to 100 for individual storm events. Although this variation appears to be related to feedlot manure pack moisture dynamics insufficient data exists to validate these claims. A simple transport capacity model was then calibrated to erosion data available from rainfall simulator studies of feedlot erosion. The transport capacity model was linked to a feedlot runoff-flow routing model to predict solids concentrations in feedlot runoff events. Modeled sediment concentrations were compared to those measured several production feedlots to validate the model with results generally indicating good agreement between measured and modeled average solids concentrations. The developed model was then used to assess the impact of various feedlot design characteristics, including feedlot area, aspect ratio, and slope, on solids transport from the feedlot surface. Overall the results indicated that minimizing feedlots slope (~2%) was important for limiting the erosive potential of feedlot runoff. Moreover, limiting pen-to-pen drainage paths and instead routing runoff water to conveyance structures will reduce flow rates limit the loss of sediment from the feedlot surface. Finally, although less frequent scrapping of concrete may have the potential to reduce runoff volumes and therefore solids transport, we recommend that solids are scraped prior to the precipitation event, especially events where more than 1 cm of precipitation is probable.

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Chapter 5. Phosphorus Retention, Accumulation, and Movement in Six Vegetative Treatment Areas on Iowa Feedlots

Abstract. *Increased environmental awareness has prompted the need for improved feedlot runoff control. Vegetative treatment systems (VTSs) provide a cost effective option that may enhance environmental security. Vegetative treatment systems are typically designed on the basis of hydraulic performance, which may result in over-application of nutrients, especially nitrogen and phosphorus. This study assessed the retention, accumulation, and movement of phosphorus in vegetative treatment areas used for runoff control on six Iowa feedlots over a four year period. Phosphorus loadings and retention were calculated based on measured settled feedlot effluent, or vegetative infiltration basin effluent, and vegetative treatment area runoff volumes and phosphorus concentrations. Results indicated that between 61 and 89% of all applied phosphorus was retained (defined as inflows minus surface outflows divided by inflows) within the treatment area, resulting in phosphorus loadings of 124 to 358 kg P/ha-yr. Measurements of harvested vegetation phosphorus concentration and yield indicated that between 13 and 61 kg P/ha-yr were removed with vegetation harvest, which accounted for only 6 to 13% of all applied phosphorus. Projected soil phosphorus accumulation was compared to annual measurements of soil Mehlich-3 phosphorus concentrations increases. Both approaches found similar increases in soil phosphorus levels, indicating that the majority of the phosphorus retained in vegetative treatment areas was due to interaction and retention in the surface soil, presumably in labile pools. Deep soil sampling (0 to 122 cm) was utilized to evaluate vertical phosphorus movement through the soil profile. Sampling indicated that most phosphorus accumulation was in the surface soil, but that signs of vertical transport and leaching were occurring after four years of operation especially near the VTA inlet.*

Keywords. Phosphorus, feedlot runoff, mass balance, soil phosphorus, Mehlich-3 phosphorus

INTRODUCTION

Open-lot animal feeding operation (AFO) runoff has been recognized as a potential pollutant to receiving waters because it contains nitrogen, phosphorus, organic matter, solids, and pathogens. The U.S. Environmental Protection Agency (EPA) developed a set of effluent limitation guidelines (ELGs) that described the design and operating criteria for feedlot runoff control systems on concentrated animal feeding operations (CAFOs) (Anschutz et al., 1979). These effluent limitation guidelines historically required collection, storage, and land application of feedlot runoff; however, recent modifications allowed the use of alternative treatment systems when the performance of the alternative systems, based on the mass of nutrients released, was equivalent to or exceeded that of an appropriately sized and managed containment system (EPA, 2006). Vegetative treatment systems (VTSs) are one possible alternative runoff control technology that has been proposed.

A VTS is a combination of treatment components, at least one of which utilizes vegetation, to manage runoff from open lots (Koelsch et al., 2006). Vegetative treatment areas (VTAs) and vegetative infiltration basins (VIBs) are two possible treatment components for VTSs. A vegetative treatment area is a band of planted or indigenous perennial vegetation situated down-slope of cropland or animal production facility that provides localized erosion protection and contaminant reduction (Koelsch et al., 2006). As vegetative treatment technology has matured different types of treatment systems have developed; examples include sloped, level, pumped, and sprinkler vegetative treatment areas and vegetative infiltration basins (Bond et al., 2011). Briefly, a sloped VTA is an area level in one dimension with a slight slope along the other, to facilitate sheet flow, planted and managed to maintain a dense stand of vegetation (Moody et al., 2006). Operation of a sloped VTA consists of applying solid settling basin effluent uniformly across the top of the vegetated treatment area and allowing the effluent to sheet-flow down the slope, whereas a level VTA uses a flood effect to distribute the effluent over the VTA surface. A pumped VTA has the increased flexibility of allowing the treatment area to be located upslope of the cropland or animal production facility, but still relies on flow to distribute effluent over the length of the vegetative treatment area surface. A sprinkler VTA has the same location flexibility as a pumped VTA, but has the additional advantage of ensuring uniform effluent application over the treatment area surface. Ikenberry and Mankin (2000) identified several possible methods in which effluent was treated by VTAs, including settling solids, infiltrating the runoff, and filtering of the effluent as it flowed through the vegetation. Additionally, interactions between the soil and soil fauna and the flowing effluent could provide additional mechanisms of nutrient retention. A VIB is a flat area, surrounded by berms, planted to perennial vegetation. A VIB uses a flood effect to distribute effluent over the surface. These areas have drainage tiles located 1 to 1.2 m (3.4 to 4 ft) below the soil surface to encourage infiltration of effluent. The tile lines collect effluent that percolates through the soil profile. The effluent then receives additional treatment, often through use of a VTA. Nutrient and pathogen removal in the VIB relies on effluent filtration as it percolates through the soil, plant uptake and harvest, degradation of the nutrients and pathogens by soil fauna, and sorption of contaminants to soil particles.

Two design approaches, one utilizing a hydraulic balance and the other a nitrogen balance, have been proposed for sizing VTAs (Woodbury et al., 2006). Previous work by Woodbury et al. (2005) has shown that if designed using the nitrogen balance approach, VTSs can successfully utilize applied nitrogen. However, in many cases VTSs have been designed based on hydraulic performance. This typically resulted in smaller VTSs, which may cause deep percolation of runoff water below the root zone and over-application of nutrients, especially nitrogen and phosphorus (Woodbury et al., 2006). As VTSs rely heavily on the soil-plant system to filter nutrients and contaminants, there is a need to understand the impacts that phosphorus application in excess of agronomic demand has on soil quality, phosphorus mobility, and the ability of the plant-soil system to retain future phosphorus applications. Research has shown that continued application of phosphorus in amounts greater than crop needs cause an accumulation of phosphorus in soil surface horizons (Sui et al., 1999). Moreover, this phosphorus accumulation is often associated with concentrated animal feeding operations and repeated manure application (Sharpley et al., 1984) and has the potential to lead to losses of soluble phosphorus in surface runoff (Sharpley, 1995; Pote et al., 1996; Pote et al., 1999). Additionally, increases in soluble phosphorus in soil drainage and in subsurface horizons have been reported (Smith et al., 1995; Eghball et al., 1996; James et al., 1996); however, because of the high phosphorus fixing capacity of most soils, vertical movement and leaching of phosphorus through the soil profile is usually low. Even under these conditions (soils with large capacities to fix and stabilize phosphorus) phosphorus leaching could occur due to both preferential flow (which would limit interaction between the soil matrix and the applied effluent) or through colloid facilitated transport.

The issues of phosphorus accumulation and fate are especially relevant in land application waste management systems where the build-up in soil phosphorus can be a major factor affecting the life expectancy of the system (Hu et al., 2006). One method of estimating the phosphorus treatment life expectancy of land application waste management systems is to determine the soil's maximum phosphorus sorption capacity and use this to estimate the amount of phosphorus the soil could potentially retain (Hu et al., 2006). This approach is based on observations that when material containing phosphorus is applied to soil, the soluble forms of phosphorus decrease with time (Holford et al., 1997), preventing losses of soluble phosphorus in runoff and leaching to groundwater but also reducing plant availability (Sui and Thompson, 2000). It has been suggested that this added phosphorus can be immobilized by organic matter, adsorbed (or absorbed) by soil particles, or react with other ions in the soil to form precipitates (Hu et al., 2006). This phosphorus sorption equation-mass balance approach was used by Baker et al. (2010) to estimate probable life expectancies of four VTA's in Iowa and to evaluate the impact of different design and management strategies for increasing the design life; however, this work did not compare the model to monitored phosphorus accumulation patterns.

The objectives of this work were to (1) evaluate if a mass balance approach was capable of predicting changes in soil test phosphorus concentrations on six vegetative treatment areas in Iowa, (2) evaluate changes in soil test phosphorus concentrations with time, and (3) use deep soil cores to evaluate if leaching or vertical redistribution of applied phosphorus was occurring. Although not a rigorous mass balance, since leaching of soluble phosphorus was not monitored, the approach utilized here still provides valuable insight into phosphorus accumulation patterns within the soil profile (both with depth in the soil profile and down the length of the vegetative treatment area) and provides insight into the fate of applied phosphorus.

METHODS AND MATERIALS

Site Descriptions

Six vegetative treatment systems were monitored as part of this study. These treatment systems were located on concentrated animal feeding operation (CAFO) sized open beef feedlots throughout the state of Iowa. The sites were described in detail in Andersen et al. (2009) and are only briefly discussed here. Data summarizing the characteristics of the Iowa State University (ISU) monitored portions of the feedlots and VTSs is provided in Table 1. Information shown includes the feedlot capacity, the VTS configuration, the size of the drainage area (feedlot and additional contributing area, if applicable), the volume of the settling basin, the area of the VIB (where applicable), and the area of the VTA.

Central Iowa 1 (CN IA 1) was a 3.09 ha feedlot permitted for 1,000 head of cattle. Runoff effluent drained into a solid settling basin designed to hold 4,300 m^3 of effluent. The VTA consisted of two channels operated in parallel; each channel was 24 m wide and averaged 311 m long. Central IA 1 VTA soil consisted of Clarion loam, Cylinder loam, and Wadena loam (Soil Survey Staff, NRCS USDA, 2010). The VTS at Central Iowa 2 consisted of a SSB, VIB, and VTA. Runoff from the 1.07 ha feedlot drained into a concrete SSB which released effluent into a 0.32 ha VIB. Effluent captured in VIB tiles was pumped onto a 0.22 ha VTA. Soils in the VIB consisted of Nicollet loam and Webster clay loam and the VTA was Harps loam (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 1 (NW IA 1) consisted of a 2.91 ha feedlot permitted to hold 1,400 head of cattle. Feedlot runoff was collected in a SSB with a maximum containment volume of 3,700 m^3. The SSB outlet pipe discharged onto VTA consisting of Galva silty clay and Radford silt loam soils (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 2 (NW IA 2) had an SSB-VIB-VTA system designed to control runoff from a 2.96 ha concrete feedlot. A settling basin collected the feedlot runoff and released it to a 1.01 ha VIB drained by 15 cm diameter perforated tiles installed 1.2 m deep and spaced 4.6 m apart. Flow from the tile lines was collected in a sump and pumped onto the VTA divided into two 27 m wide channels. The channel receiving effluent was switched manually by the producer, i.e., only one channel was loaded with effluent at a given time. Northwest IA 2 soils consisted of Moody silty clay loam (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 1 (SW IA 1) was a 7.49 ha feedlot with an 11,550 m^3 solid settling basin that released effluent to a 4.05 ha VTA divided into ten channels. Tile lines, installed to control water table depth below the system and enhance infiltration of effluent into the soil, surrounded each of the VTA channels. Soils in the VTA consisted of mostly Judson silty clay loam and smaller areas of Colo-Ely complex (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 2 (SW IA 2) was a 3.72 ha feedlot. Runoff drained into a solid settling basin and was released to a 3.44 ha VTA constructed with earthen berm spreaders perpendicular to the direction of flow along the length. The spreaders slowed the flow of effluent through the system, increasing the time for infiltration. Southwest IA 2 VTA soil consisted of Kennebec silt loam (Soil Survey Staff, NRCS USDA, 2010).

Table 1. Summary of the feedlot capacity, system configuration, and component sizes for vegetative treatment systems at each of the six sites. SSB – solid settling basin, VIB – vegetative infiltration basin, VTA – vegetative treatment area.

Site	No. of Cattle	VTS Components	Drainage Area (ha)	SSB (m^3)	VIB (ha)	VTA (ha)
Central Iowa 1	1,000	1 SSB - 2 VTA	3.09	4,290	--	1.49
Central Iowa 2	650	1 SSB - 1 VIB - 1 VTA	1.07	560	0.32	0.22
Northwest Iowa 1	1,400	1 SSB - 1 VTA	2.91	3,710	--	1.68
Northwest Iowa 2	4,000	1 SSB - 1 VIB - 1 VTA	2.96	1,120	1.01	0.60
Southwest Iowa 1	2,300	1 SSB - 10 VTA	7.49	11,550	--	4.05
Southwest Iowa 2	1,200	1 SSB - 1 VTA	3.72	6,275	--	3.44

Development of Soil P Prediction Methodology

A mass balance approach, based on the analysis presented in Baker et al. (2010), was used to predict soil phosphorus concentrations and these results were compared to soil phosphorus monitoring done for this study. The full mass balance equation is presented as Eq. 1.

$$\Delta Soil_{phosphorus} = P_{applied} - P_{runoff} - P_{vegetation} - P_{leached} \tag{1}$$

In this equation, all terms are expressed in kg of phosphorus per hectare of the vegetative treatment area. $P_{applied}$ is the mass of phosphorus applied to the vegetative treatment area from solid settling basin or vegetative infiltration basin effluent, P_{runoff} is the amount of phosphorus lost from the vegetative treatment area due to overland flow releases from the VTA, $P_{vegetation}$ is the mass of phosphorus removed by harvesting vegetation, $P_{leached}$ is the amount of phosphorus lost with percolating water (assumed to be negligible in phosphorus mass balance), and $\Delta Soil_{phosphorus}$ is the change in the amount of phosphorus stored in soil profile depth that was sampled. This can be related to changes in soil phosphorus concentration by Eq. 2.

$$\Delta Soil_{phosphorus} = \rho_b d\left(C_P - C_{P,i}\right) \qquad (2)$$

In this equation ρ_b is bulk density of the soil (kg/m^3), d is the depth of soil sample monitored (m), C_p is the concentration of phosphorus in the soil (mg P/kg soil), and $C_{p,i}$ is the background concentration of soil phosphorus before use of the vegetative treatment area (mg P/kg soil). Combining the two equations allows direct estimate of the change in soil phosphorus concentration as shown in Eq. 3.

$$\Delta C_p = \frac{P_{applied} - P_{runoff} - P_{vegetation} - P_{leached}}{\rho_b d} \qquad (3)$$

Use of this mass balance equation required monitoring of inflows and outflows from the vegetative treatment area, phosphorus concentrations in these flows, sampling of the harvested vegetation for both phosphorus concentrations and yields to determine the amount removed with harvest, and an estimate of the amount of phosphorus lost due to leaching below the zone of interest. In this analysis, the amount of phosphorus lost due to leaching was assumed to be negligible (zero) since phosphorus is usually strongly retained in the soil (although preferential flow or colloid facilitated transport could cause some losses). This assumption will be investigated by looking at the results of deep soil cores that measured phosphorus changes to a depth of 122 cm and by evaluating potential leaching losses by performing a water balance. All other parameters were monitored in this study, including changes in soil phosphorus. Monitoring methods are described below.

Monitoring Effluent Flows and Phosphorus Concentrations to and from the VTA

VTS monitoring data at CN IA 1, CN IA 2, NW IA 1, and NW IA 2 were collected from June 2006 through December 2009. Data collection at SW IA 1 and 2 began in fall and spring of 2007 respectively and ended in December 2009. The data collected included daily precipitation depths, effluent volumes released from each VTS component, and the effluent concentrations for multiple parameters (ammoniacal-nitrogen five-day biochemical oxygen demand, chemical oxygen demand, chloride, pH, total phosphorus, total Kjeldahl nitrogen, total suspended solids, nitrate-nitrogen, dissolved reactive phosphorus, and total dissolved solids). Complete descriptions of the monitoring methodologies can be found in Andersen et al. (2009).

Precipitation was measured using an ISCO 674 tipping-bucket rain gauge (Teledyne ISCO, Lincoln, NE). A passive rain gauge installed on site was used to ensure rainfall data accuracy. Iowa Environmental Mesonet data (http://mesonet.agron.iastate.edu/) were used to determine precipitation depths for events occurring between 1 November and 1 April, generally snowfall.

The effluent monitoring method used at the settling basin was dependent on outlet design. An ISCO 750 low-profile area-velocity sensor (Teledyne ISCO, Lincoln, NE.) was used at settling basins with pipe outlets. An ISCO 720 submerged probe (Teledyne ISCO, Lincoln, NE.) in conjunction with a 0.45 m (1.5 ft) H-flume was used to monitor outflow for non-pipe outlet locations. In 2006, the settling basins were passively managed and one sample was collected and sent for analysis from each SSB release. If the release continued for more than one day, an additional sample was collected for each additional day. In 2007, the producers at CN IA 1, CN IA 2, NW IA 1, SW IA 1, and SW IA 2 began actively managing SSB releases (NW IA 2 began managing SSB releases in 2008). When the SSB outlet was actively managed, the producers released small amounts of SSB effluent on consecutive days. Collecting one sample per day of SSB release proved expensive; to reduce sampling cost, a new sampling protocol was developed. This was to collect a SSB sample from the first SSB release after a rainfall event; samples from the following two days were archived in a freezer. On the third day an additional sample was sent for analysis.

At sites with a VIB, the effluent captured in the tile lines was collected in a sump and pumped onto the VTA. The pumped volume was measured using a Neptune 5 cm (2 in.) turbine flowmeter (Neptune Technology Group, Tallassee, AL). An ISCO sampler was interfaced with the turbine meter with an ISCO 780 Smart 4-20 analog interface module (Teledyne ISCO, Lincoln, NE). This allowed the amount of effluent applied to the VTA to be calculated on a daily basis. Samples were collected and shipped following the protocol described for managed SSBs.

Flow monitoring at the VTA outlet was accomplished using similar methods as those at the settling basin outlet. An ISCO 750 low-profile area-velocity sensor (Teledyne ISCO, Lincoln, Neb.) was used on sites where the VTA had a pipe outlet, and an ISCO 720 submerged probe (Teledyne ISCO, Lincoln, Neb.) in conjunction with a 0.45 m (1.5 ft) H-flume was used on the other VTAs. One sample was collected per day of release.

The mass of each parameter released during each event was calculated by multiplying the measured sample concentration and the monitored flow volume. If a representative sample was not collected for a release event, the geometric average (Andersen et al., 2009) for that year and component was substituted. The yearly mass of each parameter released was the sum of the event release totals. Calculated mass release data was then used to determine yearly reductions in contaminant mass transport and VTA phosphorus loadings.

Vegetation Sampling for Phosphorus

Vegetation sampling was conducted to determine the mass of phosphorus removed by harvesting vegetation. Vegetation samples were collected within a week of harvest. This was done by collecting vegetation samples near the inlet, outlet, and every 61 m (200 feet) down the length of the treatment area (mirroring the protocol used for soil sampling). A random spot near each sampling location was selected for sampling. A 0.093 m^2 (1 ft^2) area was harvested by cutting the vegetation 2.5 cm above the soil surface. The sample was dried to constant moisture (approximately 72 hours) in a convection oven at 35°C. Yield was calculated based on the total dry mass of sample collected and the area sampled. The dry sample mass was recorded to provide an estimate of yield which was verified by comparing to producer recorded masses of the harvested round bales. If values didn't agree within 10% the producer's harvested mass was used instead (this needed to be done if weather conditions at the site didn't allow harvesting of all biomass). The sample was ground to pass a 0.5 mm sieve using a Thomas Model 4 Wiley Mill (Thomas Scientific, Swedesboro, NJ). A 2-gram subsample of the ground sample was sent to the Iowa State University Soil & Plant Analysis Laboratory for analysis of total phosphorus and total nitrogen. Multiplying the measured phosphorus concentration of the harvested biomass times the yield provided an estimate of phosphorus removal.

Soil Sampling for Phosphorus

Surface Soil Sampling

Surface soil samples were collected on an annual basis in the fall and once before system operation began; these samples represent soil in the zero to thirty centimeter (zero to 12 inch) depth range. A surface soil sample was collected near the VTA inlet, the VTA outlet, and every 61 m (200 feet) along the length of the VTA. At each sampling location, 10 soil cores from a radius of 3 m (10 feet) around the sample location were collected and composited to make one sample. Each sample location was marked with GPS coordinates so the same location could be subsequently sampled in following years allowing change in soil nutrient content with time to be tracked. The soil samples were delivered to the ISU Department of Agronomy Soil Laboratory where they were tested for Mehlich-3 phosphorus content.

Statistical analysis of the soil phosphorus data was performed using SAS version 9.2 software (SAS Institute Inc., Cary, NC). Analysis was performed separately for each site. The analysis was run as a block design, using sample location as the blocking variable and year as the fixed factor. The year x sample location interaction term was used as the error term to test for differences in average concentration. Fisher's protected least significant difference test was used for mean comparisons.

Soil bulk density was monitored once at each site in the summer of 2007. At each site three 7.62 cm (3 inch) diameter by 46 cm (18 inch) soil cores were collected. The soil core was dried to constant weight at 105°C (approximately 24-36 hours). Soil bulk density measurements from all three cores were averaged to determine the bulk density of the VTA soil. This bulk density was assumed constant throughout all four years of monitoring. Although bulk density most likely did vary with time, the overall fluctuation was most likely small enough to have minimal impact on the results. For instance, a change in density of 0.1 g/cm^3 change would only change the estimated amount of phosphorus in the soil by approximately 6%.

The viability of the mass balance method for understanding changes in soil phosphorus concentrations was evaluated by regressing the measured change in soil phosphorus concentrations against the projected change in soil phosphorus concentration as determined using Eq. 3. This was performed in two ways. In the first method the annual change in phosphorus concentration was compared to the projected change for that year. In the second method the projection was ran cumulatively, that is the change in monitored phosphorus concentration was always compared to the baseline (pre-system operation) soil phosphorus concentration and regressed against the change the cumulative amount of phosphorus application would have caused. The advantage of the second method is that it provides a larger range of values in which to test the methodology and the impact of any errors present in estimates of phosphorus loading, removal with vegetation harvest, and in soil phosphorus concentrations was minimized since these errors get progressively smaller in comparison to the underlying trend in the data.

Deep Soil Sampling

Deep soil sampling was conducted prior to system operation and then again after 2.5 and 3.5 years of system operation. A deep soil sample was collected near the VTA inlet and also near each VTA outlet of each VTA. Each sample location was marked with GPS coordinates so the same location would be subsequently sampled in following years allowing change in soil nutrient content with time to be tracked. At each soil sampling location, a soil sampling probe (Giddings Machine Company, CO) was used to collect a 2.54 cm (1 inch) diameter soil core that was 122 cm (48-inches) long. The sample was then cut into segments to represent the 0-15.4 cm (0-6 inches), 15.4-30.5 cm (6-12 inches), 30.5-61 cm (12-24 inches), 61-94.4 cm (24-36 inches), and 94.4-122 cm (36-48 inches) depths. Each of these segments was put in a soil sampling bag and sent for analysis to the Soil and Plant Analysis Lab in the Agronomy Department at Iowa State University. These soil samples were analyzed for Mehlich-3 phosphorus.

RESULTS AND DISCUSSION

Vegetation Sampling

As discussed previously, vegetation harvest and removal is critical for vegetative treatment system sustainability as it offers the only acceptable means of phosphorus removal from the treatment area. Both the yield and the amount of phosphorus removed with the vegetation exhibited substantial variation among the six sites (table 2). This variability was often related to the number of cuttings the producer was able to harvest each year. In most cases the ability of the producer to harvest the vegetation was related to weather and soil conditions present within the vegetative treatment area. For example, vegetation was only harvested one year at Central Iowa 2; this treatment system utilized a vegetative infiltration basin that drained continuously and kept a large portion of the small vegetative treatment area in a saturated or nearly saturated condition making harvest difficult. At the remaining sites vegetation was harvested either once or twice a year as weather and soil conditions permitted. In certain instances, the VTA was only partially harvested during some of the cuttings as certain areas were too wet to support the harvest equipment. The yields of these six sites ranged from 5.29 to 14.8 Mg/ha (table 2). These yields are similar to those suggested by a University of Minnesota extension pamphlet on Reed canarygrass where 13.4 to 15 Mg/ha yields are reported for two-cut forage system near St. Paul, Minnesota (Sheaffer et al., 1990). The lower yields reported from the VTAs may be related to the opportunity for only one harvest in some years or from the fact that the systems were managed to optimize runoff disposal, not forage yield.

In general the amount of phosphorus removed trended with the mass of biomass harvested, exhibiting a significant correlation (r = 0.95). However, this wasn’t always the case as Northwest Iowa 1 tended to have lower phosphorus removals than the higher amounts of biomass removed would suggest. The vegetation at this site tended to be dominated by smooth bromegrass rather than the Reed canarygrass that dominated the other sites. This data suggests that bromegrass is less able to utilize excess phosphorus than the Reed canarygrass. This is supported by the conclusion of Kovar and Claassen (2009) that Reed canarygrass is more effective than bromegrass in depleting the soil solution of phosphorus under high phosphorus input conditions. Similarly, Sheaffer et al. (2008) found that Reed canarygrass had significantly greater phosphorus uptake than bromegrass when grown in a potato-process wastewater land application area. In their study, Sheaffer et al. (2008) found that under the potato processing wastewater application that Reed canarygrass phosphorus uptake was 31 kg P/ha while smooth bromegrass uptake averaged 25 kg P/ha. Based on this evidence, Reed canarygrass appears to better vegetation choice when excess phosphorus application is probable, such as in vegetative treatment systems.

Table 2. Yields and phosphorus removal from harvesting vegetative treatment area vegetation for Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2). NA – Not applicable; system operation began in 2007.

		CN IA 1	CN IA 2	NW IA 1	NW IA 2	SW IA 1	SW IA 2
2006	Yield (Mg/ha)	6.25	0	9.21	5.29	NA	0
	P Removal (kg/ha)	24	0	28	16	NA	0
2007	Yield (Mg/ha)	7.15	8.26	14.8	6.79	7.22	8.34
	P Removal (kg/ha)	28	29	44	21	36	32
2008	Yield (Mg/ha)	9.34	0	12.3	4.49	3.55	12.5
	P Removal (kg/ha)	36	0	38	13	16	56
2009	Yield (Mg/ha)	13.3	0	11.4	7.7	8.45	14.4
	P Removal (kg/ha)	56	0	36	26	37	61
Total P Removed (kg/ha)		144	29	146	76	89	149
% of Applied P Removed		13	6	10	6	13	16

In a second experiment, Sheaffer et al. (2008) found that phosphorus uptake could be greatly enhanced by increasing the yield of the Reed canarygrass through supplemental addition of nitrogen fertilizer. By adding supplemental nitrogen they were able to increase phosphorus uptake to 72 kg P/ha. Their results for phosphorus uptake are similar to those found in this study without supplemental nitrogen application as we monitored phosphorus uptake of up to 61 kg P/ha. The feedlot runoff vegetative treatment areas received substantially higher nitrogen application rates (593-1866 kg N/ha-yr) than the potato processing wastewater application sites (250 kg N/ha-yr), thus it is probable that sufficient nitrogen was available at these sites to achieve the higher yields without supplemental nitrogen application.

As discussed in Baker et al. (2010) ensuring an adequate balance between phosphorus inputs and outputs is necessary to prolong the phosphorus saturation life of the vegetative treatment area. Baker et al. (2010) suggested that between 30-60% of the annual phosphorus inputs should be harvested. At the sites monitored in this study only between 6 and 16% of the applied phosphorus was harvested. To achieve the phosphorus balance suggested by Baker et al. (2010), at a minimum these vegetative treatment areas need to be two to five times their current size or technologies that improve phosphorus retention within the settling basin need to be developed. The current imbalance between inputs and outputs indicates that the potential for rapid accumulation and vertical movement exists within these systems. As such, it is critical to monitor phosphorus within the treatment area to watch for signs of vertical transport or leaching of phosphorus.

Surface Soil Sampling

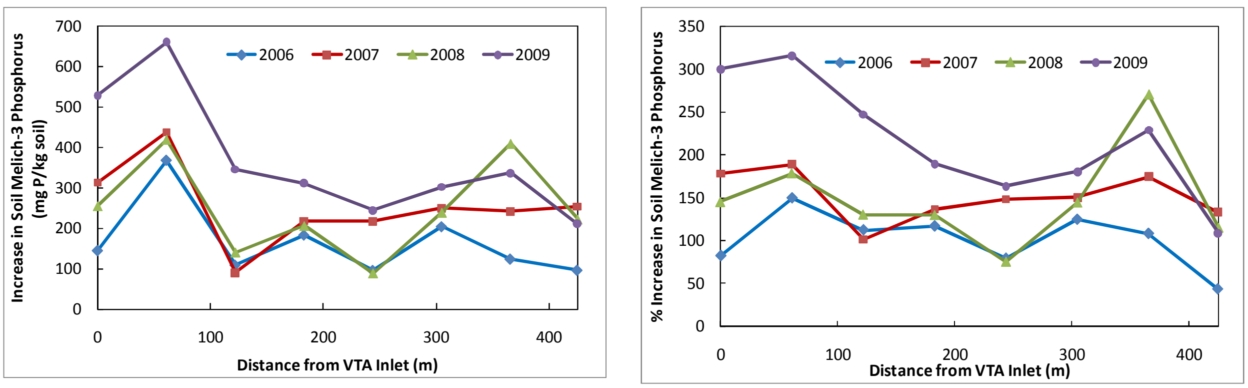
Surface soil phosphorus contents offer the first means of assessing phosphorus accumulation in the soil. Iowa soils are considered "very high" in phosphorus if the Mehlich-3 extractable phosphorus level exceeds 31 mg P/kg soil; as seen in Table 3 the background soil phosphorus concentrations at all six of these sites were well in excess of these levels (ranging from 96 to 717 mg P/ kg soil, i.e., 3-23x the Iowa very high threshold). Although these values may seem high, it needs be recognized that this recommendation on Mehlich-3 phosphorus is interpreted as the amount phosphorus at which a yield increase wouldn't be expected from additional phosphorus application, not as an index of the potential for phosphorus to affect nearby water bodies, i.e., these soil tests serve agronomic, not environmental purposes. However, elevated soil test P concentrations are often associated with increased transport.

Table 3 provides average Mehlich-3 phosphorus concentrations for each of the vegetative treatment areas before application of feedlot runoff commenced and then annually (sampled in November) thereafter. In general, all sites responded similarly with phosphorus levels increasing quickly and significantly after application of feedlot runoff commenced. Although this holds true as a general trend, there were expectations to this pattern; most notably at Central Iowa 2 and Southwest Iowa 1 which both exhibited significant increases in phosphorus concentration during their first year of operation, but then stabilized around their new phosphorus level. At Central Iowa 2 this occurred because phosphorus removal within the vegetative infiltration basin substantially limited phosphorus inputs to the VTA especially during the second through fourth year of system operation; however, no obvious explanation exists as to why levels in the Southwest Iowa 1 stabilized. Both Central Iowa 1 and Northwest Iowa 1 exhibited strong patterns of increasing phosphorus concentrations with significant differences in soil phosphorus concentrations occurring every one or two years of system operation; however, within each site there were anomalies to this pattern. Specifically, the phosphorus concentration at Central Iowa 1 showed a significant decrease in 2009 as compared to 2008 levels while at Northwest Iowa 1 2008 concentrations were similar to 2007 levels. Reasons for these exceptions are not clear, but they may be related sample variability, to conditions that were conducive to vertical transport of phosphorus deeper in the soil profile, concentrated flow paths within the VTA that either minimized or maximized inputs of phosphorus to the specific sampling locations prior to sampling, or in the case of 2009 at Central Iowa 1 drier conditions that allowed greater removal with vegetation harvest. At both Northwest Iowa 2 and Southwest Iowa 2 phosphorus concentrations showed a trend of increasing concentration with time with no exception for any of the years.

Table 3. Average surface soil (0-30 cm) Mehlich-3 phosphorus concentration (mg P/kg soil) for Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2). Values within a column that do not share the same letter are significantly different at the 0.05 level.

	CN IA 1	CN IA 2	NW IA 1	NW IA 2	SW IA 1	SW IA 2
Initial	286a	96a	190a	172a	132a	717a
2006	326ab	175b	355b	205ab	---	---
2007	386b	173b	442c	324bc	---	817a
2008	445c	176b	438c	354bc	200b	1040b
2009	345ab	156b	557d	451c	191b	1128b

Along with the average phosphorus concentrations it is also important to consider how the phosphorus was distributed down the length of the VTA. This is illustrated for the Northwest Iowa 1 VTA (figure 1). The plots show (a) the absolute change in soil phosphorus concentration (as compared to the initial soil sample) and (b) the relative change normalized as the percent change in soil concentration. Both plots showed a general pattern of greater accumulation within the upper portion (near the inlet) of the VTA, as was expected, since this would be the area where solids eroded from the feedlot and escaping the settling basin would be expected to settle. Additionally, smaller settling basin release events would not distribute the applied effluent over the entire VTA, but instead only load the effluent over the upper fraction of the treatment area. This trend is present in all years, but becomes especially evident in 2009. In general, the concentration increase was less as sample points progressed away from the VTA inlet; however, exceptions to this pattern did occur. Most notably the 365-m point in 2008 exhibited a significant increase in phosphorus concentration. The VTA experienced heavy hydraulic and phosphorus loading during this year which may have created concentrated flows paths and greater phosphorus transport and accumulation at this sampling location. Corrective actions by the producer to level and reduce channeling appear to have alleviated this in 2009 as phosphorus content at this point decreased, although remnants of the 2008 accumulation remain. This illustrates two things, first it is critical for the producer to watch for concentrated flow and to take corrective action as soon as possible to correct the situation, and second higher hydraulic loadings are prone to creating concentrated flows (Faulkner et al., 2011).



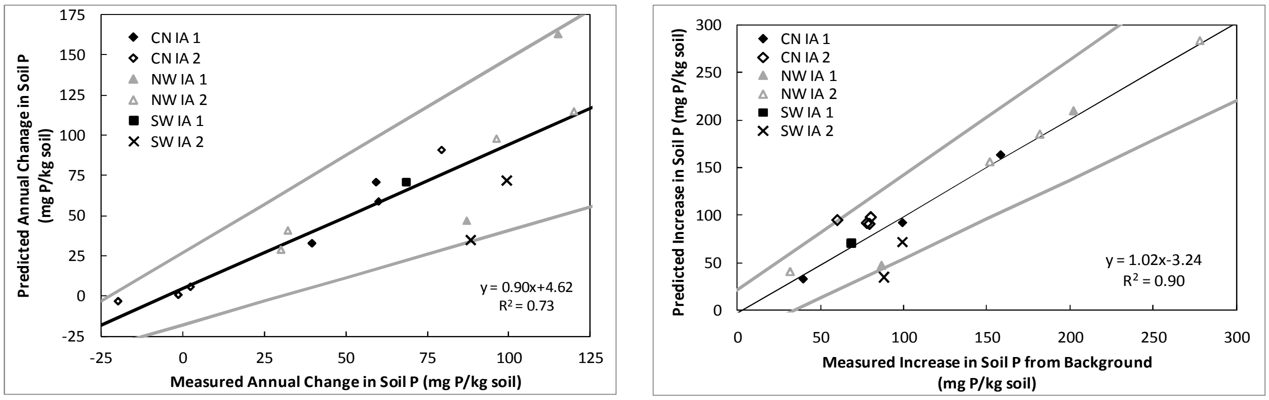
(a) **(b)**

Figure 1. Illustration of how soil Mehlich-3 phosphorus increases along the length of the Northwest Iowa 1 vegetative treatment area. (a) Increases in soil phosphorus and (b) percent increase in soil Mehlich-3 phosphorus concentration as compared to initial sample.

Comparing Predicted and Measured P Accumulation

Since the general trend in the surface soil samples was increasing phosphorus concentrations with time a direct prediction of the increase in phosphorus with the phosphorus mass balance was attempted. The predicted concentration changes were compared to the monitored change in soil phosphorus concentrations. Several soil concentration samples results were discarded to make this comparison; specifically, the Southwest Iowa 2 2008 data point was not used due to system modifications that required dirt work in the upper portion of the vegetative treatment area during modification of the system from a settling bench to a settling basin. This dirt work created a situation where the soil was removed, invalidating the mass balance approach. Additionally, the 2009 phosphorus concentration results at Central Iowa 1 were not utilized as there was a significant decrease in phosphorus content; again, dirt work at this site (modification of earthen flow spreaders within the VTA) may have been a potential cause.

In general the remaining data points showed a reasonable fit between the predicted and measured increases in soil phosphorus concentration. This is shown graphically in Fig. 2 on (a) an annual change basis and (b) on a cumulative change from background basis. In both cases the predicted change fits reasonably well with the monitored change although some discrepancies exist. These errors could be caused by numerous factors including measurement errors in the amount of phosphorus applied or lost with the runoff effluent, discrepancies in the amount of phosphorus removed with harvested vegetation, or inaccuracies in surface soil sampling. Of these three sources, non-representative soil samples are most likely as flows and concentrations were intensively monitored over the four year period and errors in overestimating loading from some events would tend to cancel with others events were loading was underestimated. Similarly although errors in sampling vegetation were certainly present, these would have minimal impact on the overall mass balance as phosphorus harvested with vegetation was a relatively small portion of the total phosphorus applied. Soil samples could be influenced by concentrated flow paths within that VTA that would result in either an under- or over- estimation of phosphorus accumulation within the soil. Although sampling at multiple points and compositing ten cores from around the sample location should help reduce the impact of soil variability and effluent channeling, it will not eliminate the impact. Additionally, soil conditions during sampling could have had some influence on results, especially if phosphorus had been recently applied. Finally, this approach assumes that no phosphorus was lost below the 30 cm sampling depth and that all applied phosphorus is in a form that is extractable by the standard Mehlich-3 extraction technique. Thus, it is possible that a high prediction of soil phosphorus could indicate that phosphorus was leached below the 30-cm depth or indicate that some of the applied phosphorus was fixed as a form not extracted by the Melich-3 procedure; however, research has indicated that larger percentages of applied P remains as soil test phosphorus in soils with high phosphorus contents. Alternatively, a low prediction would tend to indicate non-uniform phosphorus application that results in concentrated application around the sampling points (such as could be caused by flow spreaders that caused effluent ponding on sampling points).



(a) **(b)**

Figure 2. Comparison of soil phosphorus accumulation predicted using the mass balance approach and the phosphorus accumulation monitored shown on (a) annual basis and (b) cumulative basis. Grey lines represent 95% confidence intervals on the regression.

Both methods of analysis, i.e., annual and cumulative, provided reasonable agreement between the monitored and predicted change in soil phosphorus. For both model fits the y-intercept of the best fit line was not significantly different than zero and the slope was not significantly different than one at 95% confidence interval. This indicates that the mass balance method does an adequate job of predicting monitored changes in soil phosphorus. The cumulative data model fit was slightly better (R^2 = 0.90) than the annual change methodology (R^2 = 0.73); this conceptually makes sense as the cumulative analysis is able to utilize a larger range of data and the impact of the sampling errors would tend to be minimized by comparing to the larger change in soil phosphorus concentration.

Deep Soil Sampling

The final part of this investigation focused on if phosphorus was migrating vertically through the soil profile. This was done by collecting deep soil samples and analyzing them for increases in Mehlich-3 phosphorus concentration and was supplemented by an analysis of phosphorus leaching potential. Results of the deep soil samples are illustrated graphically for four sites (Central Iowa 1, Northwest Iowa 1, Northwest Iowa 2, and Southwest Iowa 1) in Figure 3. The analysis shows the average phosphorus concentration as a function of depth in the soil profile. This allows a view of the concentration front, which can be examined for both increases in concentration and for movement vertically in the soil profile. All the sites showed similar trends, increasing phosphorus concentrations near the soil surface, which is in agreement with the surface soil sampling results, and that the phosphorus front is slowly migrating vertically through the soil profile. The movement of the phosphorus front appears to become more pronounced in the fourth year of system operation. This would seem to support the hypothesis of Baker et al. (2010) that the soil will reach its phosphorus saturation limit and then phosphorus will migrate vertically through the soil. In most cases phosphorus at lower depths (greater than 0.3 meters) appears to not yet have been affected by phosphorus applications. The migration of the saturation front appears to have not yet occurred at Southwest Iowa 1. This was the last system to come into operation; thus it is probable that sufficient amount of phosphorus has not been applied to saturate the soil profile.

As discussed previously in the surface soil sampling section the distribution of phosphorus along the vegetative treatment area is also important. In this case we were most interested if this saturation and vertical transport was only occurring near the inlets of the vegetative treatment areas or if it was also occurring near the outlet. An example of the typical pattern of phosphorus concentration profiles in the vegetative treatment areas is shown in Fig. 4 for the Central Iowa 1 vegetative treatment area in 2009. It is clear from Fig. 4 that most of the leaching and phosphorus accumulation occurs near the inlets of the vegetative treatment area. As discussed previously, this is expected for gravity flow vegetative treatment areas as the effluent loading is not expected to be uniform. The inlet samples show high levels of phosphorus accumulation and are already showing signs of phosphorus leaching while the outlet samples show a much lower amount phosphorus accumulation. One other interesting feature of the figure is the degree to which phosphorus leaching has appeared to occur near the outlet of VTA 2. A small berm was built to prevent releases from the VTA; however, this berm causes ponding and saturated condtions to occur near this deep soil sampling point. It appears that these periods of saturaiton may be facilitating vertical phosphorus transport without first saturating the soil with phosphorus.

The final area of phosphorus dynamics explored was the potential for leaching. Neither phosphorus concentrations in soil water leachate nor amount of water leached was monitored. Thus several assumptions were required to formulate an estimate. The amount of water leached was estimated based on a water balance (shown as equation 4).

$$L = P + I - R - ET \tag{4}$$

In this equation L is the amount of water leached (m^3/ha), P and I represent inflows of water from precipitation and effluent application respectively (m^3/ha), and R is the amount of runoff from the vegetative treatment area (m^3/ha). Measurement of these three parameters was discussed in the materials and methods section. Evapotranspiration (ET) was estimated using potential evapotranspiration measurements available from the nearest weather station available on the ISU Agronomy Mesonet site and using this as an input for SPAW model simulations of the VTAs. Using this approach the average amount of water leached annually ranged from 1300 m^3/ha to 7700m^3/ha. Average leaching volumes were 4000, 5300, 3700, 7700, 1300, and 1300 m^3/ha for CN IA 1, CN IA 2, NW IA 1, NW IA 2, SW IA 1, and SW IA 2 respectively.

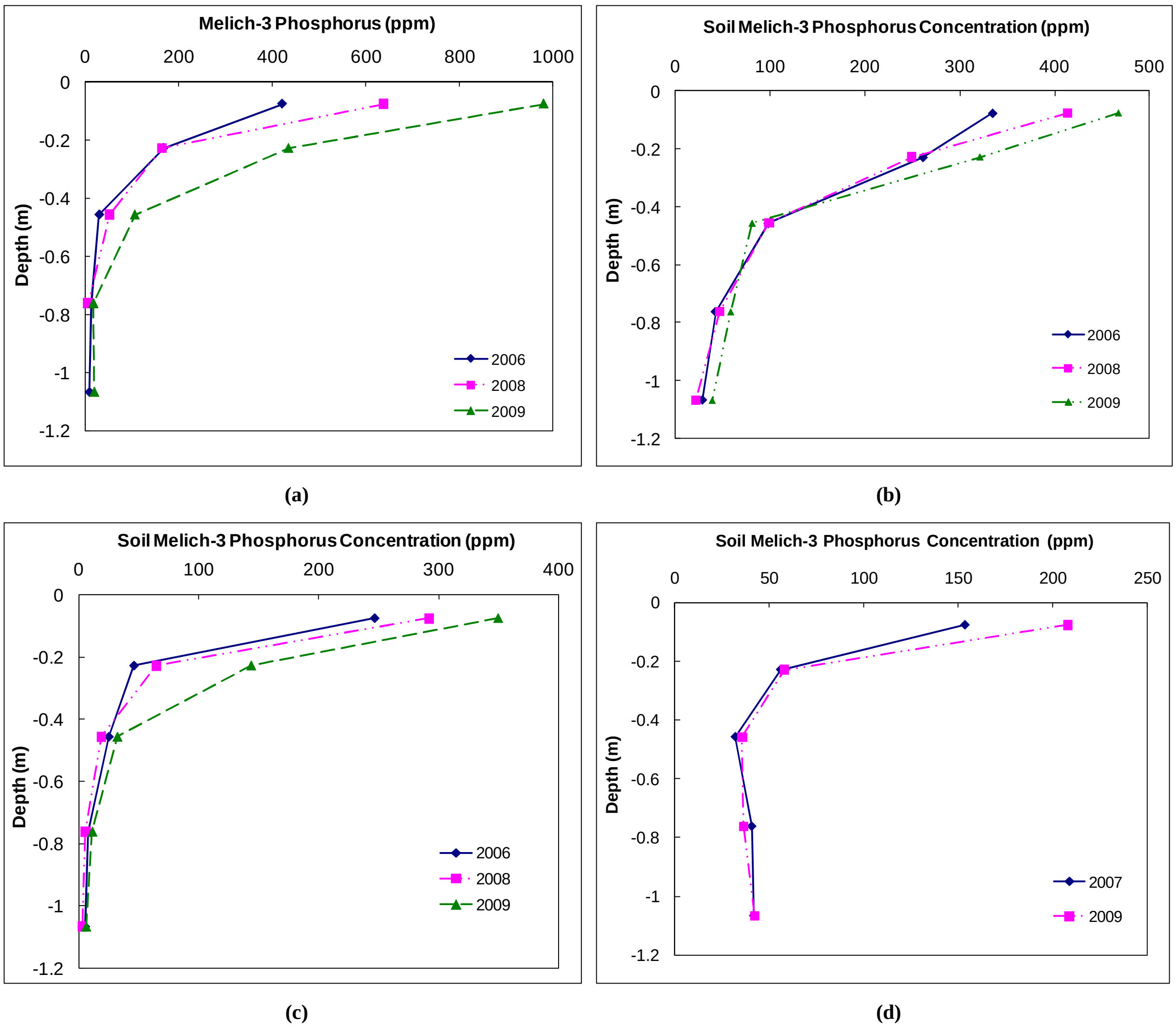


Figure 3. Mehlich-3 phosphorus concentration profiles from the surface to a depth of 1.2 m at (a) Central Iowa 1, (b) Northwest Iowa 1, (c) Northwest Iowa 2, and (d) Southwest Iowa 1.

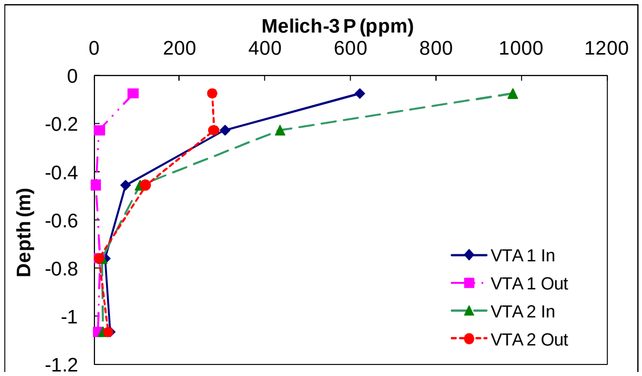


Figure 4. Comparison of Mehlich-3 phosphorus concentration profiles at vegetative treatment area inlets and outlets at Central Iowa 1 after four years of use.

Phosphorus concentrations in the leachate were not available, so they were estimated to be equal to the equilibrium phosphorus concentrations determined by Andersen et al. (2011) based on phosphorus sorption curves after five years of use as vegetative treatment areas. They found phosphorus concentrations of 1.25, 0.00, 3.82, 2.93, 0.61, and 2.15 mg P/L for CN IA 1, CN IA 2, NW IA 1, NW IA 2, SW IA 1, and SW IA 2, respectively. Assuming these concentrations are representative of drainage water from the VTA, this approach suggests that between 0 and 23 kg P/ha-yr could have potential been leached and accumulated deeper within the soil profile as observed in figure 3. Based on this analysis it appears that these systems are still retaining the majority of its phosphorus inputs in the surface soil; however, large precipitation and application events are providing sufficient flow gradient to move phosphorus deeper in the soil profile. Moreover, large events may move the effluent through the soil profile without allowing sufficient reaction time for the soil to sorb and remove phosphorus, making these leaching estimates conservative, i.e., more leaching may be occurring than is suggested here. Additionally, neither preferential flow nor colloidal transport were considered here; both of these processes would serve to increase the amount of phosphorus lost by leaching.

CONCLUSIONS

The objective of this work was to perform a preliminary phosphorus balance on six vegetative treatment systems on open beef feedlots in Iowa. The monitoring results indicated that while Reed canarygrass and bromegrass yields were substantial and capable of removing up to 60 kg P/ha it only accounted for 6-13% of the total phosphorus retained in the vegetative treatment area. This indicates that the soil plays the key role in phosphorus retention and treatment within these systems. Previous research has shown that high rates of phosphorus application can result in accumulation and eventually vertical transport of the phosphorus through the soil profile. This was confirmed in this study which showed a significant increases in Mehlich-3 extractable phosphorus concentration in the top 30 cm of the VTA soil profile, with significant increases often occurring rapidly, i.e., within one to two years of system operation. The monitored increases in soil phosphorus were strongly correlated with the mass of phosphorus applied to the VTA. Moreover, soil cores collected to a depth of 122 cm, indicated that after four years vertical transport of phosphorus was detectable to a depth of about 40 cm, although saturation had not yet occurred. This indicates that a sound design should consider a phosphorus balance prior to construction to minimize the rate at which phosphorus will accumulate within the soil and to minimize vertical transport through the soil profile. Unfortunately, specific sizing suggestions are beyond the scope of this manuscript as an economic analysis that evaluates the construction and operating costs and benefits associated with variously sized systems and includes their design life is required; however, this work shows that a phosphorus balance provides a reasonable approach to understand the rate at which phosphorus will accumulate.

Based on this analysis we provide the following suggestions for both operating and managing successful vegetative treatment systems.

- Settling basin effluent should be captured and held until after a storm event. Allowing a day or two to pass until distribution to the VTA improves performance and reduces phosphorus loading to the vegetative treatment area by allowing more time for sediment deposition. Other pretreatments that have the potential to remove phosphorus prior to application should be considered.
- Good vegetation is critical to success; this vegetation not only slows the flow and improves soil structure and infiltration, but its harvest provides the only acceptable method of phosphorus removal. Reed canarygrass appears to have greater potential for phosphorus uptake than other grasses; where possible species with high phosphorus uptake rates should be utilized.
- VTA designs should consider using multiple channels and allow the producer to determine which channels are receiving effluent. This would allow the producer to continue utilize the treatment system while being able to dry and harvest vegetation from one of the channels.
- Producers must be vigilant in watching for signs of flow channelization and maintaining uniform sheet-flow over the vegetative treatment area. Gullies and rills must be repaired by filling and reseeding the areas. This will improve hydraulic and phosphorus distribution over the VTA area.
- Soils provide the majority of phosphorus retention in the system. Selecting sites with an ability to sorb and fix large amounts of phosphorus is key to extending the life of the system.

- Methods that improve effluent distribution down the length of the VTA should be considered. Options include both sprinkler systems and surging effluent onto the VTA to distribute effluent more evenly over the length of the treatment area.

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Chapter 6. Amounts and Forms of Phosphorus in Six Iowa Grassland and Vegetative Treatment Area Soils

Abstract. *Continually land-applying manures at rates exceeding crop removal can increase soil phosphorus concentrations to levels that are of environmental concern. Information about soil phosphorus fractions is useful for understanding the bioavailability and susceptibility of phosphorus to transport. In this study, we used a sequential fractionation procedure to investigate the forms and amounts of phosphorus present in six soils used as vegetative treatment areas for controlling feedlot runoff for five years and a paired soil that did not receive the effluent application. Five soil samples were collected to a depth of 30 cm from within the vegetative treatment area and five more from the paired location at each of six sites. Soil phosphorus was then partitioned into a series of pools using a modified Hedley fractionation scheme (dividing soil P into nine empirical fractions [$CaCl_2$ soluble, $NaHCO_3$-soluble inorganic and organic P, NaOH-soluble inorganic and organic P, HCl-soluble P, a second set of NaOH-soluble inorganic and organic P, and residual P]). With the exception of one site, phosphorus concentrations were significantly higher in the vegetative treatment area soils than in the paired sample. However, the distribution of phosphorus between labile ($CaCl_2$ and $NaHCO_3$ extractable) and recalcitrant (the NaOH, HCl, and residual P) pools was not significantly affected ($p = 0.24$) by the amount of phosphorus in the soil. Regression analysis was utilized to evaluate phosphorus partitioning as related to the soil's total phosphorus content. Results indicated that inorganic pools were increasing in an approximately linear fashion with the soil's total phosphorus content, but that organic pools showed no relationship to total phosphorus. Results did not indicate saturation of any of the inorganic phosphorus pools had occurred.*

Keywords. Feedlot runoff, phosphorus, Hedley fractionation, vegetative treatment system, soil phosphorus

INTRODUCTION

Phosphorus plays a vital role in determining the productivity of terrestrial ecosystems, but if not properly managed can accumulate and be transported to aquatic environments where it can cause eutrophication. In general, soils contain between 100 and 3,000 mg P/kg soil, most of which is present as orthophosphate compounds; however, significant quantities of organic phosphorus can be present (Condron and Newman, 2011). This is indicative that soil phosphorus dynamics are often controlled by a combination of chemical and biological processes (Frossard et al., 2000); for example, soil P transformations include precipitation-dissolution and adsorption-desorption reactions which control phosphorus transfer between the solid and liquid phases as well as the biological processes of immobilization and mineralization which controls transformations between inorganic and organic forms.

Quantifying the forms and amounts of phosphorus present in soils offers one method to evaluate the effects that phosphorus amendments and land management changes have on phosphorus accumulation and mobility; however, this analysis is complicated by the interactions of phosphorus compounds within the soil matrix. Thus, instead of focusing on specific phosphorus compounds, classes of soil phosphorus are often defined functionally based on their extractability with different chemical reagents. This has lead to the development of several sequential fractionation procedures (Chang and Jackson, 1957; Hedley et al., 1982). In these procedures a soil sample is sequentially reacted with a series of reagents which are assumed to dissolve discrete sets of phosphorus compounds (Sui et al., 1999). This provides operationally defined phosphorus fractions that can then be used to interpret the impact of the amendment or management change on the status of the soil phosphorus, such as plant availability or susceptibility to losses (Tiessen and Moir, 1993).

Sequential extraction techniques offer a means in which to evaluate impact of land management changes on phosphorus distributions. One land use that has received increased scrutiny in recent years is land application of animal manures. In the last 30 years, crop and livestock operations have become increasingly specialized, resulting in spatially separated production systems (Kellogg et al., 2000). This intensification and specialization has been driven by greater demand for animal products and improved profitability, and has caused a transfer of phosphorus from grain-producing areas to animal-producing regions, resulting in localized phosphorus surpluses and increased soil phosphorus concentrations (Sharpley et al., 2004; Lanyon, 2000). In addition to the direct impacts of phosphorus addition, manure can increase soil pH (Eghball, 200; Kingery et al., 1994) due to input of large amounts of Ca and the buffering effects of bicarbonates and organic acids present or released during manure decomposition (Sharpley and Moyer, 2000; Sharpley et al., 2004). These modifications to soil properties can alter the forms of phosphorus present in the soil, impacting phosphorus stability.

Due to increased environmental concerns, open lot cattle operations have become more conscientious about preventing release of contaminated runoff from their operation to water bodies; however, due to the relatively dilute nature of feedlot runoff, redistribution for land application at agronomic rates can be expensive. This has led increased interest in other alternatives that can be utilized to control and treat the feedlot runoff. Vegetative treatment systems (VTSs) are one possible alternative runoff control technology that has been proposed. A VTS is a combination of treatment components, at least one of which utilizes vegetation, to manage runoff from open lots (Koelsch et al., 2006). Vegetative treatment areas (VTAs) and vegetative infiltration basins (VIBs) are two possible treatment components for VTSs. A vegetative treatment area is a band of planted or indigenous perennial vegetation situated down-slope of cropland or animal production facility that provides localized erosion protection and contaminant reduction (Koelsch et al., 2006). Operation of a VTA consists of applying solid settling basin effluent uniformly across the top of the vegetated treatment area and allowing the effluent to sheet-flow down the slope. Two design approaches, one utilizing a hydraulic balance and the other a nitrogen balance, have been proposed for sizing VTAs (Woodbury et al., 2006); however, both design procedures can result in phosphorus application in excess of agronomic demand.

As VTSs rely heavily on the soil-plant system to filter nutrients and contaminants, there is a need to understand the impacts that phosphorus application in excess of agronomic demand has on the quantity and forms of phosphorus within the soil. Thus, the objectives of this study were to (1) document the effects that five years of vegetative treatment system operation had on the amount and forms of phosphorus present in the top 15-cm of the soil profile at six vegetative treatment aresa, (2) to evaluate how the distribution of phosphorus, i.e., the percent of phosphorus in various fractions, differed between the vegetative treatment area and paired grassland area soils at each site, and (3) to investigate if the distribution of phosphorus was related to the soils total phosphorus content. Hypotheses were developed for each of the objectives; for objective 1 we hypothesize that the large additions of manure would result in greater phosphorus content in most of the inorganic pools in the vegetative treatment area soils, but only small increases in organic phosphorus contents. This pattern of accumulation was anticipated because we expected that the already carbon rich topsoil would accumulate relatively little carbon during the five years of effluent application, resulting in little to no increase in microbial biomass or soil organic matter (Stewart et al., 2007). Large increases in inorganic phosphorus were anticipated based on phosphorus sorption experiments that indicated that large quantities of phosphorus (~500 to 1000 mg P/kg soil) could be sorbed by these soils during 24-hour incubations. For objective 2 we hypothesized that vegetative treatment soils would have larger percentages of their phosphorus in stable sodium hydroxide and hydrochloric acid extractable pools than their grassland counter parts. This response was anticipated based on ancillary evidence that showed the vegetative treatment area soils’ reaction were more basic than their grassland counterparts; this would result in decreased calcium phosphate solubility and greater precipitation of calcium phosphate and phosphates associated precipitated calcium carbonates. This hypothesis is supported by the work of Sharpley et al. (2004) who demonstrated that water extractable phosphorus content tended to become a progressively smaller percentage of the soils Mehlich-3 phosphorus content at higher phosphorus levels, i.e., greater amounts of phosphorus were stored in pools of greater stability. Finally, for objective 3, we anticipated a nonlinear relationship between NaOH extractable phosphorus and soil total phosphorus content. We predicted that at lower phosphorus contents the NaOH pool would increase rapidly due to iron and aluminum fixation of the applied phosphorus; however at high phosphorus levels this phosphorus pool would saturate, resulting in little to no response with further increases in total phosphorus. When this occurred we expected calcium phosphate, i.e., hydrochloric acid extractable phosphorus, to be the major pool for stable phosphorus accumulation. This hypothesis is based on the iron and aluminum to have a limited capacity to fix phosphorus, as supported by phosphorus sorption experiments. We then assume no limit to the calcium phosphate pool since that application of feedlot runoff would continually add calcium and phosphorus to the soil-solution system.

METHODS AND MATERIALS

Site Descriptions

Six vegetative treatment systems were located on concentrated animal feeding operation (CAFO) sized open beef feedlots throughout the state of Iowa and intensively monitored over a four year period by Iowa State University. The sites were described in detail in Andersen et al. (2009) and are only briefly discussed here. Data summarizing the characteristics of the Iowa State University (ISU) monitored portions of the feedlots and VTSs are provided in table 1. Information shown includes the maximum cattle capacity of the feedlot, the VTS configuration, the size of the drainage area (feedlot and additional contributing area), the volume of the settling basin, the area of the VIB (where applicable), and the area of the VTA. Characteristics of the sites are discussed below.

Table 1. Summary of the system configuration and vegetative treatment system components at each site.

Site	No. of Cattle	VTS Components	Drainage Area (ha)	SSB (m^3)	VIB (ha)	VTA (ha)
Central Iowa 1	1,000	1 SSB - 2 VTA	3.09	4,290	--	1.49
Central Iowa 2	650	1 SSB - 1 VIB - 1 VTA	1.07	560	0.32	0.22
Northwest Iowa 1	1,400	1 SSB - 1 VTA	2.91	3,710	--	1.68
Northwest Iowa 2	4,000	1 SSB - 1 VIB - 1 VTA	2.96	1,120	1.01	0.60
Southwest Iowa 1	2,300	1 SSB - 10 VTA	7.49	11,550	--	4.05
Southwest Iowa 2	1,200	1 SSB - 1 VTA	3.72	6,275	--	3.44

Central Iowa 1 (CN IA 1) was a 3.09 ha feedlot permitted for 1,000 head of cattle. Runoff effluent drained into a solid settling basin designed to hold 4,290 m^3 of effluent. The VTA consisted of two channels operated in parallel; each channel was 24 m wide and averaged 311 m long. Central IA 1 VTA soil consisted of Clarion loam, Cylinder loam, and Wadena loam (Soil Survey Staff, NRCS USDA, 2010). The VTS at Central Iowa 2 consisted of a SSB, VIB, and VTA. Runoff from the 1.07 ha feedlot drained into a concrete SSB which released effluent into a 0.32 ha VIB. Effluent captured in VIB tiles was pumped onto a VTA. Soils in the VIB consisted of Nicollet loam and Webster clay loam and the VTA was Harps loam (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 1 (NW IA 1) consisted of a 2.91 ha feedlot permitted to hold 1,400 head of cattle. Feedlot runoff was collected in a SSB with a volume of 3,700 m^3. The SSB outlet pipe discharged onto VTA consisting of Galva silty clay and Radford silt loam soils (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 2 (NW IA 2) had an SSB-VIB-VTA system designed to control runoff from a 2.96 ha concrete feedlot. A settling basin collected the feedlot runoff and released it to a 1.01 ha VIB drained by 15 cm diameter perforated tiles installed 1.2 m deep and spaced 4.6 m apart. Flow from the tile lines was collected in a sump and pumped onto the VTA divided into two 27 m wide channels. The channel receiving effluent was switched manually by the producer. Northwest IA 2 consisted of Moody silty clay loam (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 1 (SW IA 1) was a 7.49 ha feedlot with an 11,550 m^3 solid settling basin that released effluent to a 4.05 ha VTA was divided into ten channels. Tile lines, installed to control water table depth below the system and enhance infiltration of effluent into the soil, surrounded each of the VTA channels. Soils in the VTA consisted of mostly Judson silty clay loam and smaller areas of Colo-Ely complex (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 2 (SW IA 2) was a 3.72 ha feedlot. Runoff drained into a solid settling basin and was released to a 3.44 ha VTA constructed with earthen berm level spreaders along the length. The spreaders slowed the flow of effluent through the system, increasing the time for infiltration and promoting sedimentation of particulates suspended in the flow. Southwest IA 2 VTA soil consisted of Kennebec silt loam (Soil Survey Staff, NRCS USDA, 2010). At each site grass areas of the same soil series were found and sampled to evaluate soil phosphorus content and distribution of soil not receiving the effluent application; these properties are thought to represent the original site conditions prior to use of the vegetative treatment system, and thus provide an opportunity to evaluate the impact of five years of runoff effluent application on soil phosphorus.

Soil Sampling and Analysis

At each of the six sites five soil samples were collected from the vegetative treatment area and five more from a paired area (located on the same soil series, planted to grasses, located near animal feeding facility so subjected to the same climate) that did not receive the feedlot runoff effluent application. This sampling methodology was utilized as soil sample collected before vegetative treatment construction and use were not available. Each soil sample was collected by compositing soil from five randomly selected locations within the vegetative treatment area or paired area; at each sampling location a 1.27-cm (0.5 inch) diameter push-probe was used to collect soil to a depth of 15.2 cm (6 inches) from twenty spots within a 1.5-m radius of the selected location. This sampling methodology was used to minimize the within treatment component variability due to differences in greater phosphorus loading near settling basin inlets and variability in soil properties over the relatively large vegetative treatment areas. Collected soil was placed in a plastic bag, placed on ice, and brought back to the Agricultural Waste Management Lab at Iowa State University. Once back the soil samples mass was determined and they were spread out on trays to air dry. Aggregates were crushed and sieved to pass a screen with 2 mm openings. Rocks and visible vegetation were removed during the sieving process. The mass of soil passing and retained on the 2 mm screen was determined to estimate the amount of course fraction present in each soil and determined the moisture content of the soil. A subsample of the soil passing the 2 mm screen was dried in an oven at 105°C for 24 hours to determine the air dried moisture content of the soil. The remaining soil was placed in screw-cap plastic bottles and stored until use in the sequential fractionation procedure.

Sequential Fractionation of Soil Phosphorus

A modification of the methods of Hedley et al. (1982) was selected in this study to extract empirically defined pools of phosphorus. A 0.5-g air dried, < 2 mm soil sample was placed in a 50-mL centrifuge tube and was sequentially extracted with 30 mL each of 0.01 M $CaCl_2$ solution, 0.5 M $NaHCO_3$ (pH = 8.2) solution, 0.1 M NaOH solution, 1 M HCl solution, and a second NaOH fraction. Modifications made to the Hedley procedure included not using an anion exchange membrane with the 0.01 M $CaCl_2$ extraction. The resin wasn't used since it wasn't required to obtain detectable phosphorus quantities and the phosphorus extracted without the resin was thought to be a better representation of what could potentially be transferred to surface runoff. Additionally, the second NaOH extraction was performed after the HCl extraction. The extraction was added based on the review of Condron and Newman (2011) and the method of Tiessen and Moir (1982) in which the authors suggest a second NaOH extraction, following acid extraction, can extract a significant fraction of the remaining organic phosphorus. The use of a second NaOH extraction is consistent with long-established methodologies for extracting soil organic matter and allows a more detailed characterization of phosphorus that is often lumped into the residual pool. All other extractions followed the standard Hedley procedure and resulting fractions were defined according to the procedures outlined by Hedley et al. (1982) and Tiessen and Moir (1993). The $CaCl_2$ solution was used to extract what was assumed to be the most labile inorganic phosphorus and was presumed to be closely related to the phosphorus content of surface runoff. The bicarbonate extracted phosphorus was also considered labile phosphorus that was thought to be adsorbed on the surfaces of more crystalline sesquioxides or carbonates in the soil (Mattingly, 1975). Organic phosphorus extracted by the bicarbonate was considered easily mineralizable and to contribute to plant available phosphorus (Bowman and Cole, 1978). The hydroxide-extractable P is thought to have lower plant availability and be associated with amorphous and less crystalline Al and Fe oxides. The acid extraction is thought to extract all calcium phosphates present in the soil, including the appetite. The second hydroxide extraction was used to extract organic phosphorus that Tiessen and Moir (1993) speculate may have participated in relatively short-term biolobical transformations. The remaining phosphorus, i.e., the residual phosphorus, is considered to be the most stable in the soil, but which is part of an unidentifiable pool.

The centrifuge tubes, filled with the soil and extracting solution, were placed on their side and shaken for 16 hours on an orbital shaker (200 rpm). The soil solution was then centrifuged at 8,000*g* for 10 minutes at room temperature (24°C). The supernatant was decanted and filtered through a 0.45-μm membrane filter. The solution was neutralized (pH between 6 and 8) using dilute (1 N followed by a 0.1 N back titration) NaOH or HCl and phenolphthalein indicator. A portion of the filtered and neutralized solution was then analyzed for molybdate reactive phosphorus using the ascorbic acid method. Sample absorbance was measured on a Genesys 6 spectrophotometer at a wavelength of 880 nm. This portion was termed inorganic P. Another portion of the same extract was oxidized using a sulfuric acid-nitric acid digestion (4500-P B4, Standard methods for examination of water and wastewater), neutralized using phenolphthalein indicator and dilute NaOH, and then analyzed for molybdate reactive phosphorus using the ascorbic acid method. The difference between the total phosphorus determined on the digested solution and the inorganic phosphorus was termed organic phosphorus. Organic phosphorus contents were determined for the $NaHCO_3$ and both the NaOH solutions. After the sequential extraction, the residual soil P was determined using a nitric acid-perchloric acid digestion with neutralization of the digest again occurring before phosphorus was determined using the ascorbic acid method.

An analysis of variance was conducted using SAS version 9.2 software (SAS Institute Inc., Cary, NC) to evaluate statistical differences in the amount of phosphorus in each of the phosphorus fractions ($CaCl_2$ soluble, $NaHCO_3$-soluble inorganic and organic P, NaOH-soluble inorganic and organic P, HCl-soluble P, a second NaOH-soluble inorganic and organic P, and residual P). The statistical model used consisted of Site, Application History (VTA or Grass), a Site*Application History interaction, and replication nested within the Site*Application History interaction. Contrast statements were used to determine if within site differences in soil phosphorus contents existed.

RESULTS AND DISCUSSION

The main objective of this study was to document how the large amounts of P added to the soil was distributed into the various fractions of soil phosphorus and to evaluate if incorporating these large amounts of phosphorus had altered the distribution of phosphorus between labile and recalcitrant pools within the soil. The results will first focus on the absolute concentrations of phosphorus and how they differed between the vegetative treatment area soil and the grassland area soil with a specific interest in which pools accumulated the most phosphorus. The relative concentration, i.e., the percent of phosphorus in different pools, will then be evaluated to determine if the accumulation of phosphorus has altered the distribution of phosphorus in the soil and how the availability of phosphorus may have been impacted.

The amounts of phosphorus measured in each of the pools are provided in Table 2. In general, all of the paired grassland soils that did not receive the runoff application had relatively consistent total phosphorus contents, ranging from 1,042 to 1,386 mg P/kg soil. The vegetative treatment area soils had more variable phosphorus contents ranging from a low of 760 mg P/kg soil at Central Iowa 2 to a high of 2,518 mg P/kg soill at Northwest Iowa 1. At four sites (Central Iowa 1, Northwest Iowa 1, Northwest Iowa 2, and Southwest Iowa 2) the vegetative treatment area soil total phosphorus concentrations were significantly higher ($\alpha = 0.05$ level) than the soil from the paired grassland area. At Southwest Iowa 1 VTA soil phosphorus concentration was on average higher than their grassland counterpart, but the difference was not significant ($p = 0.15$). Central Iowa 2 showed the opposite trend; at this site the paired soil sample that didn’t receive the effluent application had significantly higher phosphorus concentrations than the vegetative treatment area soil. This site utilized a vegetative infiltration basin to treat feedlot runoff prior to application of the effluent onto the vegetative treatment area. This treatment methodology resulted in the application of large amounts (between 33 and 86 cm annually) of relatively dilute (average Total P less 5.5 mg/L) wastewater. In the previous four years, i.e., prior to collection of the vegetative treatment area soil samples, total phosphorus applications were approximately equal to phosphorus losses in vegetative treatment area runoff. All other sites applied significantly more phosphorus to the vegetative treatment area than was removed in the harvested vegetation or lost in runoff from the vegetative treatment area. Thus, at sites that received phosphorus inputs in excess of agronomic demand phosphorus levels were significantly greater in the vegetative treatment area soil than in the paired grassland soil (p-value = 0.0043 from contrast statement at the five sites that received phosphorus application in excess of agronomic demand [Central Iowa 1, Northwest Iowa 1, Northwest Iowa 2, Southwest Iowa 1, and Southwest Iowa 2]).

Looking at these results in more detail we see that at Central Iowa 1 there was no difference in the most labile phosphorus ($CaCl_2$ extractable) between the VTA and the paired grassland soils. However, the majority of the other phosphorus fractions at this site ($NaHCO_2$-IP, $NaHCO_3$-OP, NaOH-IP, HCl-IP) were significantly higher in the VTA soil than in the non-amended soil from the paired grassland location. Most notably, the inorganic sodium bicarbonate, the inorganic sodium hydroxide, the organic sodium hydroxide, and the hydrochloric acid extractable pools, were all enriched by 100 mg P/kg soil or more as compared to the grassland soil. Northwest Iowa 1 and 2 and Southwest Iowa 1 and 2 showed similar trends to those of Central Iowa 1 with all the inorganic ($CaCl_2$, $NaHCO_3$, NaOH, and HCl extractable) phosphorus pools having large and significant increases in phosphorus as compared to their grassland counterparts. Central Iowa 2 showed trends opposite of this; presumably again do to the lower phosphorus loading rate utilized at this location.

The organic phosphorus pools ($NaHCO_3$-OP, NaOH-OP, NaOH-OP 2, Residual P) tended to exhibit much more variable responses with no statistical difference between the vegetative treatment area and the paired soil sample typically occurring. If differences were detected they typically occurred in the more easily extractable $NaHCO_3$ organic phosphorus pool than in the more stable NaOH-OP pools. Likewise, the residual phosphorus pools were relatively similar for vegetative treatment area soils and their paired counterparts. In general, these results support our original hypothesis that increases in phosphorus content would occur mostly in inorganic pools since the organic matter content of the soils would remain relatively unchanged due to their high organic matter content.

Along with the overall amounts of phosphorus in each pool, the distribution, i.e., the percent of total phosphorus in each pool, was also investigated and is presented in Figure 1. Although certain sites experienced changes in phosphorus distribution, for instance Northwest Iowa 1 VTA soil had a greater percentage of its phosphorus in the inorganic sodium bicarbonate extractable pool than the paired grassland soil at this site; no consistent trend was evident across all sites. These results do not support our original hypothesis that a greater percentage of the phosphorus will be available in more labile ($CaCl_2$ and $NaHCO_3$ extractable) pools. The absolute size of these pools often were considerably larger in the VTA soil than in the paired grassland soil, but they were not consistently a larger portion of the total phosphorus content. Likewise, the acid extractable (calcium phosphates) pool sizes were typically larger in the vegetative treatment area than in the paired soil, but again their proportion of the total phosphorus content in VTA soil was similar to that of the paired grassland soil. Based on these results it did not appear that phosphorus was selectively accumulating in specific pools, but rather all pools accumulated phosphorus at roughly proportional rates.

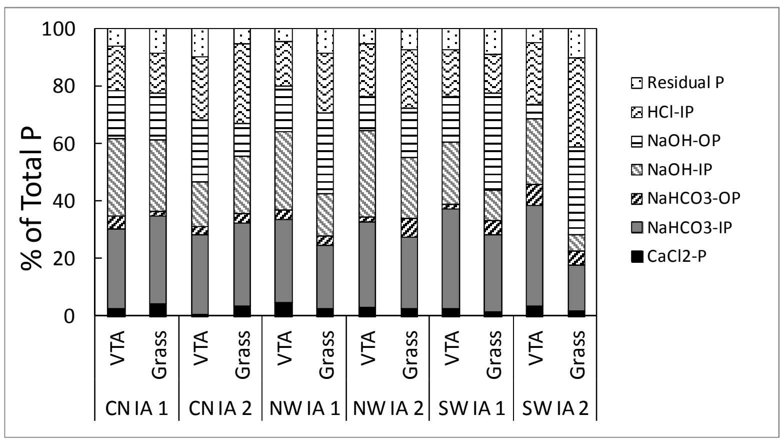

Figure 1. Comparison of phosphorus distribution in vegetative treatment areas (VTA) soils and their paired grassland counterparts at the six sites Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2)].

Table 2. Absolute concentrations of soil phosphorus in each of the soil fractions for the vegetative treatment area (VTA) and the paired soil area (Paired) at each of the six sites [Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2)]. IP – Inorganic phosphorus; OP – organic phosphorus

		$CaCl_2$-IP (mg/kg)	$NaHCO_3$-IP (mg/kg)	$NaHCO_3$-OP (mg/kg)	NaOH-IP (mg/kg)	NaOH-OP (mg/kg)	HCl-IP (mg/kg)	NaOH-IP (mg/kg)	NaOH-OP (mg/kg)	Residual P (mg/kg)	Total P (mg/kg)
CN IA 1	VTA	56	607	102	570	261	343	22	104	139	2205
	Paired	55	422	25	320	159	192	22	72	119	1386
	p-value	0.952	*0.009*	*0.035*	*0.012*	0.117	*0.002*	0.909	*0.008*	*0.050*	*<0.001*
CN IA 2	VTA	4	211	20	110	120	168	8	45	75	760
	Paired	46	398	43	256	78	381	19	80	77	1379
	p-value	*<0.001*	*<0.001*	0.082	*<0.001*	0.197	*<0.001*	*0.040*	*0.023*	0.691	*<0.001*
NW IA 1	VTA	117	719	83	657	260	392	31	144	115	2518
	Paired	35	301	42	181	262	284	18	123	120	1365
	p-value	*0.001*	*<0.001*	*0.014*	*0.001*	0.988	*0.018*	*<0.001*	0.056	0.598	*<0.001*
NW IA 2	VTA	60	678	41	605	207	415	74	66	125	2272
	Paired	29	330	83	262	144	263	19	82	101	1314
	p-value	*0.044*	*0.001*	*0.006*	*0.002*	0.472	*0.004*	*<0.001*	0.221	*0.031*	*0.003*
SW IA 1	VTA	30	452	24	260	153	213	18	57	97	1304
	Paired	15	301	55	114	351	154	7	33	103	1133
	p-value	*0.016*	*0.013*	0.074	*<0.001*	0.128	*0.029*	*0.010*	*0.013*	0.142	0.077
SW IA 2	VTA	62	696	146	443	39	430	14	57	100	1988
	Paired	18	165	51	52	277	323	7	41	108	1042
	p-value	*0.003*	*<0.001*	0.058	*<0.001*	*0.016*	*<0.001*	0.085	0.171	0.002	*<0.001*

A regression analysis was utilized to further investigate these results. In the analysis the amount of phosphorus in each pool was plotted against the total phosphorus content of the soil, this follows from the approach of Stewart et al. (2008) used to demonstrate that different pools of soil organic carbon had saturation capacity. We had hypothesized a nonlinear relationship between NaOH extractable phosphorus and soil total phosphorus content. We predicted that at lower phosphorus contents this pool would increase rapidly, due to iron and aluminum fixation of the applied phosphorus; however at high phosphorus levels this phosphorus pool would saturate, resulting in little to no further response with further increases in total phosphorus. We expected $CaCl_2$ and $NaHCO_3$ pools to accumulate a greater percentage of phosphorus once phosphorus levels are high as this would suggest that more of the phosphorus is being partitioned into more available pools. Finally, we expected HCl extractable phosphorus to increase more quickly at high phosphorus levels as the application of feedlot runoff has the potential to add calcium to the soil as well as lead to a more basic soil reaction, both of which lead to the formation of calcium phosphorus precipitates. However, we saw no evidence of these trends (Fig. 2a-d). In the case of NaOH extractable phosphorus (Fig. 2c) this may indicate that the iron and aluminum hydroxides have not yet become saturated with phosphorus. This result is supported by phosphorus sorption experiments performed on the vegetative treatment soil after five years of use that indicate VTA soils still have the ability to sorb significant quantities of phosphorus (Andersen et al. 2012). Sorption experiments do not differentiate between precipitation mechanisms and adsorption by soil particles (including oxides), but do provide evidence that a hypothesized saturation dynamic may not be occurring yet. Similarly, we did not see preferential formation of calcium phosphate precipitates at high phosphorus contents (Fig. 2d) or increasing percentages of phosphorus being stored in the $CaCl_2$ and $NaHCO_3$ pools.

Based on the results of the regression analysis it appears that several phosphorus stabilization mechanisms are occurring as phosphorus is partitioning into all the inorganic pools; however, a greater amount of this phosphorus is being partitioned into the NaOH pool (0.349 mg NaOH extractable P per mg P versus 0.134 mg HCl extractable P per mg P) perhaps indicating that sorption is a more important mechanism than calcium precipitation. These trends appear to be continuing even at the higher soil phosphorus levels, and thus do not support the saturation hypothesis. Another interesting trend was that of the $CaCl_2$ extractable phosphorus, based on the work of Sharpley et al. (2004) we had hypothesized that this pool would increases more slowly at higher soil phosphorus contents since they reported lower percentages of phosphorus was water extractable (most labile pool) with increasing Mehlich-3 soil phosphorus. We found that the most labile phosphorus pool increased in a mostly linear fashion with increasing phosphorus content; similarly, the sodium bicarbonate pool also exhibited a linear trend of phosphorus accumulation.

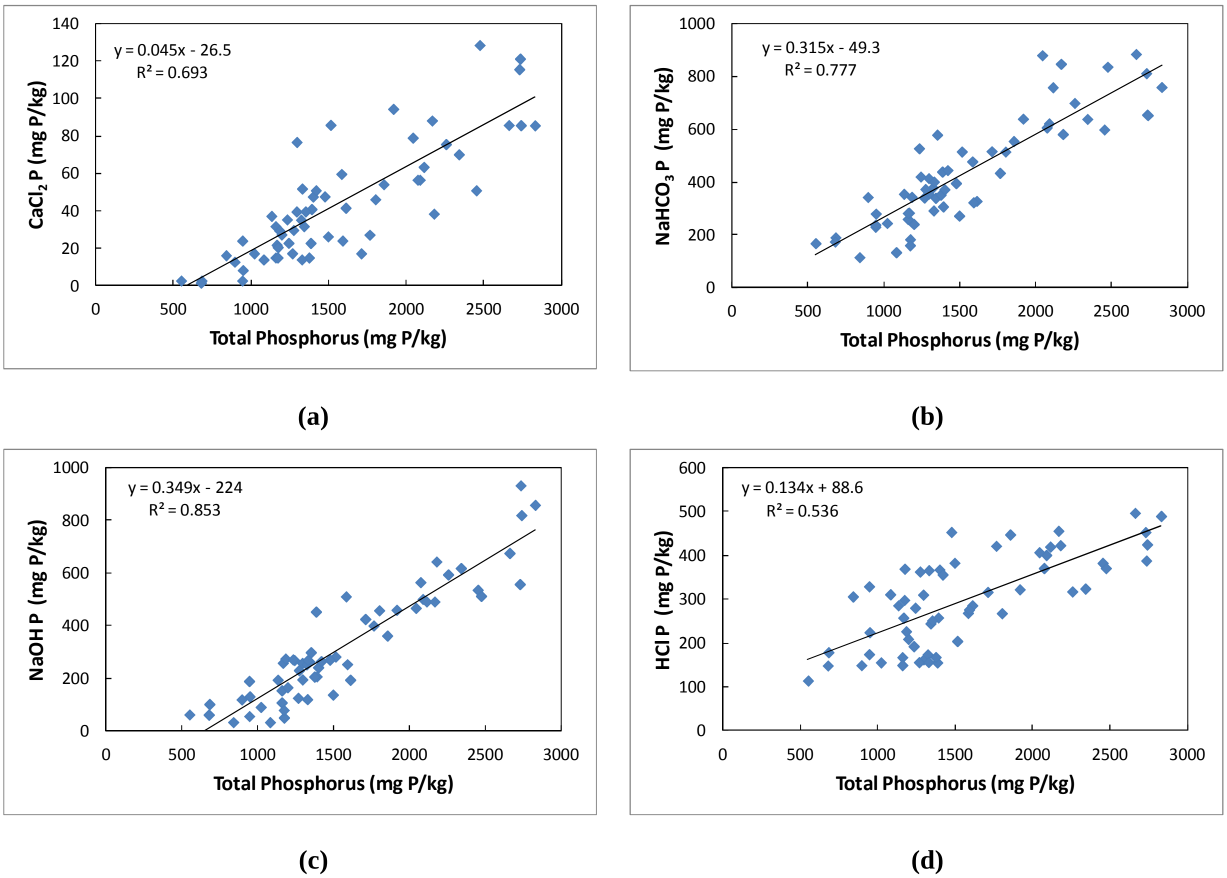

Figure 2. Relationship between (a) calcium chloride extractable phosphorus, (b) sodium bicarbonate extractable inorganic phosphorus, (c) sodium hydroxide extractable inorganic phosphorus, and (d) hydrochloric acid extractable phosphorus versus total phosphorus.

CONCLUSIONS

Phosphorus fractionation can provide considerable insight into how phosphorus is being retained in the soil and its potential for bioavailability and loss from the ecosystem. In this study we fractionated surface soil from areas that had been used as vegetative treatment areas for the previous five years and from a paired grassland counterpart at each location. Our results indicated that in general, land use as vegetative treatment areas will result in rapid accumulation of phosphorus in the surface soil and that the majority of this phosphorus will be stored in inorganic forms within the soil. Interestingly, we did not find significant changes in the distribution of phosphorus among the fractions despite increases in soil pH which were expected to promote accumulation in the calcium phosphate pool. Furthermore, although the phosphorus application had increased the sizes of the labile pools, fractionation results indicated that these labile forms were approximately 1/3 of the phosphorus in both the grassland and vegetative treatment area soils. A regression analysis indicated that the $CaCl_2$, $NaHCO_3$, NaOH, and HCl extractable inorganic phosphorus pools were responding linearly with total phosphorus content of the soil indicating that even at higher phosphorus contents the soils were not showing preferential accumulation of calcium phosphates and indicating even at these high phosphorus contents the aluminum and iron oxides in the soil still had potential to sorb and fix more phosphorus. In general, organic phosphorus contents of the soil remained consistent, presumably because the carbon contents of these already carbon-rich soils remained relatively consistent even with the addition of feedlot runoff. Overall, the results indicate that the vegetative treatment soils are converting much of the incorporated phosphorus to availabe labile forms that minimize the risk of phosphorus losses to the environment as long as perennial vegetation is maintained; however, labile forms of phosphorus are also increasing with about 1/3 of all applied phosphorus remaining in these pools.

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Chapter 7. Phosphorus Sorption Capacity of Six Iowa Soils as Affected by use as Vegetative Treatment Areas

Abstract. *Accumulation of phosphorus in soil is a major factor limiting the operational life of land application waste disposal systems. Moreover, for nutrient management purposes and evaluation of potential environmental problems it is necessary to understand the impact that manure application has on soil phosphorus sorption characteristics. In this study, laboratory experiments were conducted to investigate the impact of feedlot runoff effluent application on phosphorus sorption capacities, equilibrium phosphorus concentrations, and phosphorus buffering capacities of six Iowa soils. Soil samples were collected from vegetative treatment areas that had received feedlot runoff application for the previous five years and from a paired grassed area that did not receive the effluent application. Subsamples of each soil were incubated with a series of twelve phosphorus solutions ranging in concentration from 0 to 200 mg P/L to determine the sorption characteristics of the soil. The Langmuir sorption model was fitted to the Langmuir sorption model to determine the phosphorus equilibrium concentration, the phosphorus buffering capacity, and the maximum phosphorus sorption capacity of the soil. Sorption parameters of the paired soils were then compared to evaluate the impact effluent application had on soil phosphorus sorption properties. Results indicated that vegetative treatment areas generally had elevated phosphorus equilibrium concentrations in relation to the grassed areas, indicating an elevated risk of loss of dissolved phosphorus. In most cases the ability of the soil to sorb phosphorus was significantly increased. These results indicated that vegetative treatment area life had been extended due to increased phosphorus sink capacity of the soil; however, sizable increases in the soil solution equilibrium phosphorus content indicate that despite these increases in phosphorus sink capacity*

Keywords. Feedlot runoff, phosphorus, soil sampling, vegetative treatment system, buffering capacity, equilibrium phosphorus concentration, Langmuir sorption model

INTRODUCTION

The fate of phosphorus is one of the most critical factors for determining the sustainability, life expectancy, and effectiveness of land application waste treatment systems (Hu et al., 2006). In most land application systems, the amount of waste applied is constrained by either hydraulic or nitrogen loading considerations; this typically results in phosphorus application in excess of agronomic demand and can cause accumulation of phosphorus in the soil profile (Sui et al., 1999). This is potentially of environmental concern if the increased phosphorus levels result in greater mobility and transport to surface waters. In crop production systems this concern is typically related to the possibility of erosion and transport of phosphorus enriched soil particles. As a result many states have proposed application limits based on phosphorus indexes, which switch application constraints to phosphorus inputs when soil phosphorus levels build to a critical threshold. Similar issues exist on municipal wastewater treatment system land application areas; however, since these systems typically utilize perennial vegetation, concern over erosion and transport of particulate bound phosphorus is minimized. In these cases phosphorus application typically is not limited by soil P test levels, but is instead limited by the ability of the soil to react with the phosphorus and prevent its transport. Since most soils have high phosphorus fixing capacities the amount of phosphorus that can be applied is quite substantial; however, research has indicated that continual application of excess phosphorus, i.e., above the agronomic requirement, can change soil phosphorus chemistry and increase solubility, potentially leading to leaching or enhanced transport of dissolved phosphorus in surface runoff (Sharpley et al., 2004), and failure of the waste treatment system.

Although this disposal approach to waste treatment has generally been limited to municipalities, the increasing concentration of the animal feeding industry and the decoupling of crop and livestock production systems have prompted renewed interest in advanced treatment techniques for disposal of byproducts associated with animal production, specifically manures and process wastewaters. Although many treatment options have been suggested, most still rely on land application for final disposal due to the difficulty in meeting the stringent water quality limitations required for discharge. This has created a demand for agricultural waste management systems where the main goal is no longer to utilize nutrients for agronomic production, but instead is to minimize the costs of treating and handling the production byproducts while minimizing any pollution associated with their management and disposal. One example of this type of system is the use of vegetative treatment systems (VTSs) for feedlot runoff control. These systems provide a lower cost alternative to the traditional storage-land application system for managing feedlot runoff (Bond et al., 2011), but are designed based on a waste disposal, not nutrient utilization, paradigm.

A VTS is a combination of treatment components, at least one of which utilizes vegetation, to manage runoff from open lots (Koelsch et al., 2006). A VTS typically consists of a solid settling basin followed by either a vegetative treatment area (VTA) or a VTA in combination with a vegetative infiltration basin (VIB), although other configurations are possible. Briefly, a sloped VTA is an area level in one dimension with a slight slope along the other, to facilitate sheet flow, planted and managed to maintain a dense stand of vegetation (Moody et al., 2006). Operation of a sloped VTA consists of applying solid settling basin effluent uniformly across the top of the vegetated treatment area and allowing the effluent to sheet-flow down the slope. Ikenberry and Mankin (2000) identified several possible methods in which effluent was treated by VTAs, including settling solids, infiltration, and filtering of the effluent as it flowed through the vegetated area. A VIB is a flat area, surrounded by berms, planted to perennial vegetation. A VIB uses a flood effect to distribute effluent over the surface. These areas have drainage tiles located 1 to 1.2 m (3.4 to 4 ft) below the soil surface to encourage infiltration. The tile lines collect effluent that percolates through the soil profile and transport it to a sump, where it receives additional treatment, often through use of a VTA. Nutrient and pathogen removal in the VIB relies on effluent filtration as it percolates through the soil, plant uptake and harvest, degradation of the nutrients and pathogens by soil fauna, and sorption of contaminants to soil particles and organic matter.

Vegetative treatment systems are capable of converting applied carbon and nitrogen to gaseous forms (either aerobic or anaerobic decomposition for carbon and ammonia volatilization or denitrification for nitrogen), and thus remove them from the internal nutrient cycling of the treatment system; this doesn’t occur for phosphorus. Thus, the only environmentally acceptable method for removing phosphorus from the treatment area is via vegetative uptake and harvest; this implies that vegetative treatment systems rely heavily on the soil system to filter and retain phosphorus applications, especially when nutrients are supplied in excess of crop need. In practice, phosphorus transport is controlled in large part by the sorption behavior of the soil, which can be investigated by equilibrating soil with solutions of differing phosphorus concentrations and then evaluating how the applied phosphorus partitions between the soil solid and liquid phases. This approach is based on observations that when material containing phosphorus is applied to soil, the soluble forms of phosphorus decrease with time (Holford et al., 1997), preventing losses of soluble phosphorus in runoff and leaching to groundwater but also reducing plant availability (Sui and Thompson, 2000).

Although phosphorus sorption experiments have been widely employed for estimating phosphorus mobility in soils, relatively little work evaluating how soil phosphorus sorption capacity and sorption strength are modified by previous phosphorus application is available. Work that has been performed has been inconclusive, or at the very least site specific, indicating that in some cases (soil x manure application rate combinations) the ability of soil to sorb new additions of phosphorus has been significantly decreased while in other cases the soil’s ability to retain new phosphorus is increased. The issue of repeated phosphorus application is discussed briefly in the review of Barrow (2008), and suggests that when a nutrient, such as phosphorus, is added to a previously fertilized soil; the sorption curve will not be the same as it would have been if all the nutrient addition had occurred at once.

Further Barrow (2008) suggests understanding how the pathways of sorption are altered and the overall impact on sorption parameters are in need of greater evaluation.

The issue of phosphorus retention in soils is especially relevant to waste management systems where repeated application of phosphorus containing waste products is common. Thus, the objective of this work was to evaluate and compare phosphorus sorption patterns from paired soils that either received or did not receive application of feedlot runoff over the previous five years. The analysis was performed for six sites where soil from the vegetative treatment area and from a paired grass area was collected and the phosphorus sorption experiment was performed. Comparing the patterns from the two soils allows an evaluation of the impact that use of the vegetative treatment system had on soil phosphorus sorption properties and provides insight into how the life expectancy and performance of these waste management systems have changed.

METHODS AND MATERIALS

Site Descriptions

Six vegetative treatment systems were located on concentrated animal feeding operation (CAFO) sized open lot beef feeding operations throughout the state of Iowa and intensively monitored over a four year period by Iowa State University. The sites were described in detail in Andersen et al. (2009) and are only briefly discussed here. Data summarizing the characteristics of the Iowa State University (ISU) monitored portions of the feedlots and VTSs are provided in table 1. Information shown includes the maximum cattle capacity of the feedlot, the VTS configuration, the size of the drainage area (feedlot and additional contributing area), the volume of the settling basin, the area of the VIB (where applicable), and the area of the VTA. Conditions at each site are summarized in the following section.

Table 1. Summary of the system configuration and vegetative treatment system components at each site.

Site	No. of Cattle	VTS Components	Drainage Area (ha)	SSB (m^3)	VIB (ha)	VTA (ha)
Central Iowa 1	1,000	1 SSB - 2 VTA	3.09	4,290	--	1.49
Central Iowa 2	650	1 SSB - 1 VIB - 1 VTA	1.07	560	0.32	0.22
Northwest Iowa 1	1,400	1 SSB - 1 VTA	2.91	3,710	--	1.68
Northwest Iowa 2	4,000	1 SSB - 1 VIB - 1 VTA	2.96	1,120	1.01	0.60
Southwest Iowa 1	2,300	1 SSB - 10 VTA	7.49	11,550	--	4.05
Southwest Iowa 2	1,200	1 SSB - 1 VTA	3.72	6,275	--	3.44

Central Iowa 1 (CN IA 1) was a 3.09 ha feedlot permitted for 1,000 head of cattle. Runoff effluent drained into a solid settling basin designed to hold 4,290 m^3 of effluent. The VTA consisted of two channels operated in parallel; each channel was 24 m wide and averaged 311 m long. Central IA 1 VTA soil consisted of Clarion loam, Cylinder loam, and Wadena loam (Soil Survey Staff, NRCS USDA, 2010). The VTS at Central Iowa 2 consisted of a SSB, VIB, and VTA. Runoff from the 1.07 ha feedlot drained into a concrete SSB which released effluent into a 0.32 ha VIB. Effluent captured in VIB tiles was pumped onto a VTA. Soils in the VIB consisted of Nicollet loam and Webster clay loam and the VTA was Harps loam (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 1 (NW IA 1) consisted of a 2.91 ha feedlot permitted to hold 1,400 head of cattle. Feedlot runoff was collected in a SSB with a volume of 3,700 m^3. The SSB outlet pipe discharged onto VTA consisting of Galva silty clay and Radford silt loam soils (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 2 (NW IA 2) had an SSB-VIB-VTA system designed to control runoff from a 2.96 ha concrete feedlot. A settling basin collected the feedlot runoff and released it to a 1.01 ha VIB drained by 15 cm diameter perforated tiles installed 1.2 m deep and spaced 4.6 m apart. Flow from the tile lines was collected in a sump and pumped onto the VTA divided into two 27 m wide channels. The channel receiving effluent was switched manually by the producer. Northwest IA 2 soils consisted of Moody silty clay loam (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 1 (SW IA 1) was a 7.49 ha feedlot with an 11,550 m^3 solid settling basin that released effluent to a 4.05 ha VTA was divided into ten channels. Tile lines, installed to control water table depth below the system and enhance infiltration of effluent into the soil, surrounded each of the VTA channels. Soils in the VTA consisted of mostly Judson silty clay loam and smaller areas of Colo-Ely complex (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 2 (SW IA 2) was a 3.72 ha feedlot. Runoff drained into a solid settling basin and was released to a 3.44 ha VTA constructed with earthen berm level spreaders along the length. The spreaders slowed the flow of effluent through the system, increasing the time for infiltration and promoting sedimentation of particulates suspended in the flow. Southwest IA 2 VTA soil consisted of Kennebec silt loam (Soil Survey Staff, NRCS USDA, 2010). At each site grass areas of the same soil series were found and sampled to evaluate soil phosphorus sorption properties of soil not receiving the effluent application; these properties are thought to represent the original site conditions prior to use of the vegetative treatment system, and thus provide an opportunity to evaluate the impact of five years of runoff effluent application on sorption properties.

Soil Sampling and Analysis

At each of the six sites five soil samples were collected from the vegetative treatment area and five more from the paired grass area not receiving the feedlot runoff effluent application. Each soil sample was collected by compositing soil from five randomly selected locations within the vegetative treatment area or paired grass area; at each sampling location a push-probe was used to collect soil to a depth of 15.2 cm (6 inches) from twenty spots within a 1.5-m radius of the selected location. This sampling methodology was used to minimize the within treatment component variability due to differences in greater phosphorus loading near settling basin inlets, possible flow channelization altering nutrient distribution within treatment area, and to minimize the impact of variability in soil properties over the relatively large sampling areas. Collected soil was placed in a plastic bag, placed on ice, and brought back to the Agricultural Waste Management Lab at Iowa State University. Once back the soil samples mass was determined and they were spread out on trays to air dry. Aggregates were crushed and sieved to pass a screen with 2 mm openings. Rocks and visible vegetation were removed during the sieving process. The mass of soil passing and retained on the 2 mm screen was determined to estimate the amount of course fraction present in each soil and estimate moisture content. A subsample of the soil passing the 2 mm screen was dried in an oven at 105°C for 24 hours to determine the air dried moisture content of the soil. The remaining soil was placed in screw-cap plastic bottles and stored until use in the phosphorus sorption curve incubations.

Phosphorus Sorption Experiment

Phosphorus sorption curves were developed using the method of Graetz and Nair (2009). One gram of air-dried soil was placed into each of ten 50 mL centrifuge tubes with screw-on caps and mixed with 25 mL of 0.01 M calcium chloride ($CaCl_2$) solution containing phosphorus concentrations of 0, 1, 2, 5, 10, 25, 50, 100, 150, and 200 mg KH_2PO_4-P/L. Two additional centrifuge tubes received 0.25 g and 0.50 g of soil respectively, which were mixed with 25 mL of 0.01 M calcium chloride solution with 0 mg KH_2PO_4-P/L. These higher dilution ratio samples were added to better evaluate the response of the soil at the low phosphorus concentration range. Samples were placed horizontally on an orbital shaker and shaken end-to-end for 24 hours at 25±2°C. Samples were then placed upright and allowed to settle for one hour. The supernatant was filtered through a 0.45 μm filter. Dissolved reactive phosphorus (DRP) concentrations were analyzed spectrophotometrically at a wavelength of 880 nm using a Genesys 6 (Thermo Electron Corporation, Madison, WI) photospectrometer following the ascorbic acid method procedure (AWWA, 1998). The amount of phosphorus sorbed by the soil was calculated as the difference between the amount of phosphorus added in the original solution and the amount remaining in the equilibrated solution after 24 hours.

Phosphorus Sorption Curve Fitting

Sorption data were fitted with a modified Langmuir sorption curve (Eq. 1) as presented by Zhou et al. (2005).

$$S' = \frac{S_{\max} kC}{1 + kC} - \left(\frac{S_{\max} kC_0}{1 + kC_0} + \frac{C_0 V}{M} \right) \tag{1}$$

In this equation *S'* represents the amount of phosphorus sorbed by the soil from the applied solution (mg P/kg soil), S_{max} represents the maximum amount of phosphorus the soil can sorb (mg P/kg soil), k is a constant related to the binding energy of phosphorus to the soil (L/kg), C is the concentration of phosphorus remaining in solution after equilibration with the soil (mg P/L), C_0 is the concentration of phosphorus in solution after equilibration when the initial solution contained no phosphorus (mg P/L), V the volume of solution used in the equilibration (L), and M is the mass of soil used in the incubation (kg). As used here there were three fitting parameters in this equation, these are k, S_{max}, and C_0. In this case C_0 was used as a fitting parameter since three, rather than just one, soil samples were equilibrated with the 0 mg P/L solution. Results from equation 1 were compared against the data generated using equation 2, and measured values of initial and equilibrated solution phosphorus concentration, to determine phosphorus sorption.

$$S' = \frac{(C_i - C)V}{M} \tag{2}$$

In this equation C_i represents the initial concentration of the phosphorus solution and all remaining terms are as defined previously.

The modified version of the Langmuir sorption curve (Eq. 1) was selected because in some instances the equilibrated soil was calculated to have negative sorption, i.e., phosphorus on the soil desorbed into solution. This is typical of soils with high initial phosphorus concentrations and occurs because the true value of sorbed phosphorus (*S*) consists of both phosphorus sorbed during the incubation (*S'*) and preexisting sorbed phosphorus that is exchangeable (S_0). Mathematically, this is shown as Eq. 3.

The modified Langmuir equation accounts for this desorption of legacy phosphorus and recognizes that it is a function of the dilution ratio used in the experiment.

$$S = S_0 + S' \tag{3}$$

Equation 1 was fit to each data set, i.e., sorption data for each of the sixty soil samples, using nonlinear regression. S_{max}, k, and C_0 were iteratively adjusted to minimize the sum of the squared differences between S' calculated using Eqs. 1 and 2. This nonlinear regression was performed using the Solver function of Microsoft Excel; all parameters were allowed to vary freely except for C_0, which was required to have a value greater than or equal to zero. Based on the values of the fitted parameters, five additional terms were calculated, these were: the equilibrium phosphorus concentration (EPC_0) in mg P/L, the amount of native sorbed phosphorus (S_0) in mg P/kg soil, the soil's phosphorus buffering capacity (BC) in ([mg P/kg soil] / [mg P/L]), the remaining sorption capacity of the soil (mg P/kg soil), and the percent phosphorus saturation of the soil (%). These were calculated using Eqs. 4, 5, 6, 7, and 8 respectively.

$$EPC_0 = \frac{S_0}{k\left(S_{\max} - S_0\right)} \tag{4}$$

$$S_0 = \frac{S_{\max} k C_0}{1 + kC_0} + \frac{C_0 V}{M} \tag{5}$$

$$BC = \frac{S_{\max} k}{\left(1 + kC\right)^2} \tag{6}$$

$$SC = S_{\max} - S_0 \tag{7}$$

$$\%Sat = 100 \frac{S_0}{S_{\max}} \tag{8}$$

The EPC_0 is the solution phosphorus concentrations that causes an equal amount of sorption and desorption of phosphorus and is often interpreted as an indicator of the soluble phosphorus loss potential of the soil to both runoff and leachate. Higher values indicate a greater potential for phosphorus to be lost to runoff or drainage water (Zhou et al., 2005; Zhang et al., 2008), while low values indicate reduced phosphorus loss potential. It should be recognized that the value calculated is a function of the soil-to-solution ratio used in the study and does not directly provide *in situ* values of soil solution phosphorus concentration; however, work by Zhang et al. (2008) did suggest the this parameter was correlated with *in situ* phosphorus concentrations. This makes this parameter of great interest to water quality as it provides an assessment of soluble reactive phosphorus levels. S_0 indicates the amount of phosphorus sorbed to the soil under field conditions and provides an index to assess if use of a soil as a vegetative treatment area has increased the amount native phosphorus sorbed to the soil. The buffering capacity (BC) provides an index of the ability of a soil to resist further increases in soil solution phosphorus concentration as it provides information on how much phosphorus can be sorbed before the soil solution concentration increases by 1 mg P/L; this term is calculated based on the first derivative of equation 1. The sorption capacity (SC) provides information about how much more phosphorus could potentially be sorbed by the soil, and the percent phosphorus saturation indicates how much of the phosphorus sorption capacity is currently filled by native phosphorus.

Statistical Analysis Methods

An analysis of variance was conducted using SAS version 9.2 software (SAS Institute Inc., Cary, NC) to evaluate statistical differences in EPC_0, S_0, BC, SC, % Sat, S_{max}, and k. The statistical model used consisted of Site, Application History (VTA or Grass), a Site*Application History interaction, and replication nested within the Site*Application History interaction. Contrast statements were used to determine if within site differences in soil sorption parameters existed between the VTA and Grass land use history at each site.

RESULTS AND DISCUSSION

Analysis of Phosphorus Sorption Curves

Figures 1 a-f show the complete phosphorus sorption curves for the VTA and Grass soils at each of the six sites. Each point on a figure represents the average value of five replicate soil samples from within the VTA or the paired grass area. Error bars were added in both the x- and y-directions. Error bars in the x direction represent the standard deviation of the measured equilibrium concentration for each initial phosphorus concentration. Error bars in the y-direction represent a standard deviation of the calculated soil sorption. Also displayed in the figures is the fitted Langmuir sorption curve for each of the Site*Application combinations. All data were well fit by the equation with R^2 values greater than 0.98. All samples also exhibited a plateau to the amount of phosphorus that could be sorbed with this generally occurring at soil solution equilibrium concentration of around 80 mg P/L, providing visual support that a Langmuir type model was appropriate for analysis and interpretation of the results.

Five of the six sites showed the same general trend, greater amounts of phosphorus sorption by the VTA soil samples than the grass-area soil samples at high solution phosphorus concentrations. The only site not following this trend was SW IA 1, where the grass soil had slightly higher phosphorus sorption capacities than the vegetative treatment area soil. At low equilibrium solution phosphorus concentration most sites had greater phosphorus sorption by the grass soil samples than the VTA soils samples. The reduction of phosphorus sorption by VTA soils at low phosphorus solution concentrations was expected as the five years of use as a vegetative treatment area had drastically increased soil test phosphorus concentrations by 100 to 400 mg Mehlich-3 P/kg soil at most sites (Andersen et al., 2011). Central Iowa 2 was unique in that its vegetative treatment area soil exhibited higher sorption at low concentrations than the soil samples from the grass area. The VTA soil at this site had exhibited steady to slightly lower Mehlich-3 phosphorus concentrations over the previous three years and had a neutral phosphorus balance over this time, i.e., phosphorus additions to the VTA were approximately equal to phosphorus losses in VTA runoff. This period of steady phosphorus concentrations may have resulted in fixation and diffusion of phosphorus deeper into mineral particles and left the mineral surfaces more able to accumulate new phosphorus. Alternatively, this may have been due to phosphorus losses in leachate from VTA that washed the desorbable native phosphorus from the soil or incorporation of this labile phosphorus into the growing vegetation.

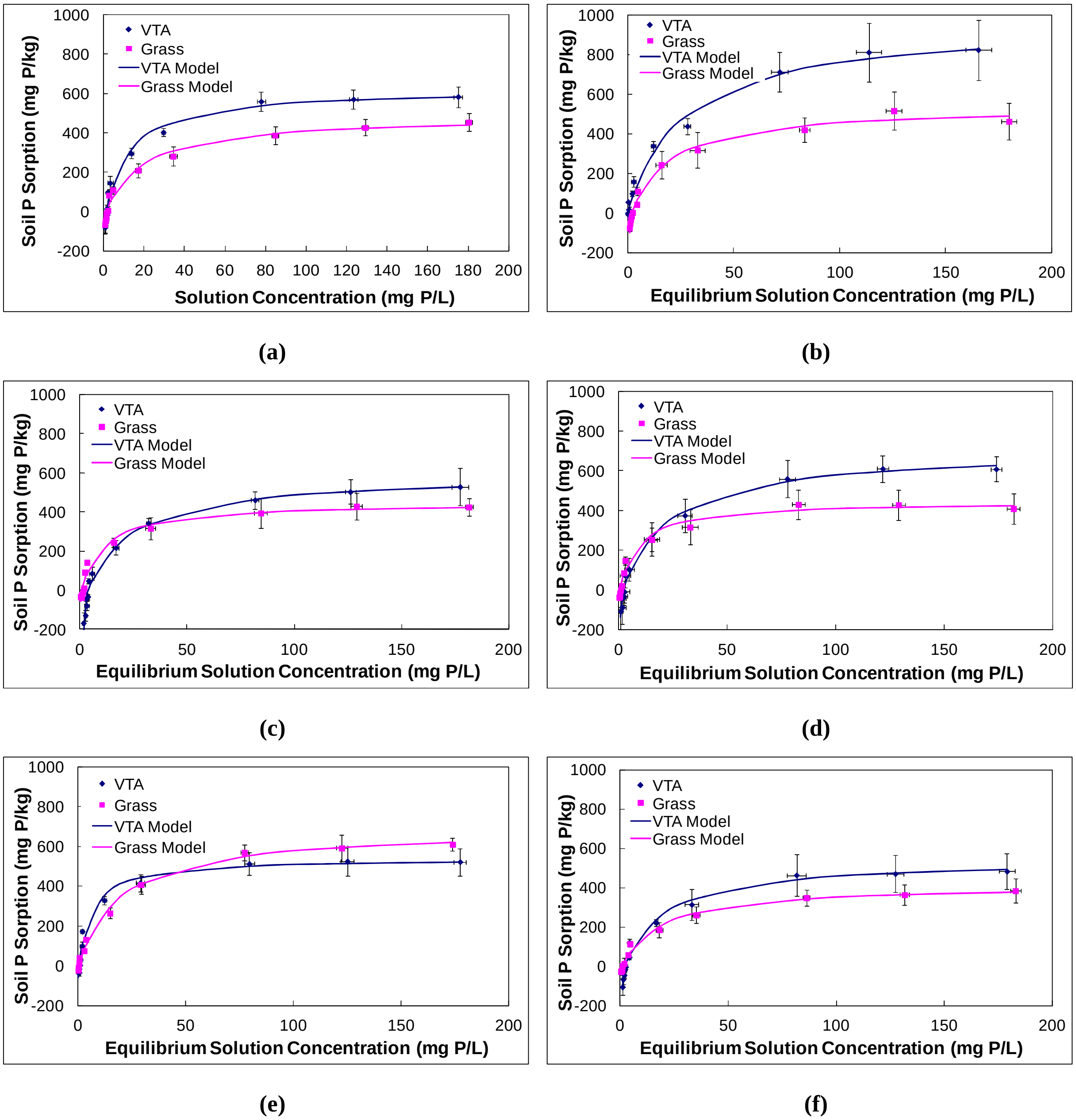

Figure 1. Phosphorus sorption curves for vegetative treatment area and grassed area soils at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, (f) SW IA 2. Each point in the figures represents the average of five soil samples. Solid lines represent model fits of the modified Langmuir equation to monitored data.

Calculated Phosphorus Sorption Parameters

Average results of phosphorus sorption properties, calculated based on the fitted Langmuir equation, are shown in Table 2. Parameters shown include the equilibrium phosphorus concentration (EPC_0), the amount of native sorbed phosphorus (S_0), the phosphorus buffering capacity (BC), the phosphorus binding energy (*k*), the maximum phosphorus sorption capacity of the soil (S_{max}), the remaining soil phosphorus sorption capacity (SC), and the percent phosphorus saturation (% P Saturation). Data were analyzed in two ways the first was to perform a site-by-site comparison between the vegetative treatment area and grassed area soil samples to evaluate if statistical differences existed. In addition to this analysis, a second analysis where data were blocked by site was performed to evaluate if results could be generalized across the sites used in this study. These results are discussed below for each parameter.

Table 2. Equilibrium phosphorus concentration (EPC_0), the amount of native sorbed phosphorus (S_0), the phosphorus buffering capacity (BC), the phosphorus binding energy (*k*), the maximum phosphorus sorption capacity (S_{max}), the remaining soil phosphorus sorption capacity (SC), and the percent phosphorus saturation (% P Saturation) of vegetative treatment area and grassed area soil samples from Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2). SEM = standard error of the mean. Bolded values indicate that the value for the VTA and Grass soils samples were statistically different ($p < 0.05$) at that site.

Site	Land Use	EPC_0 (mg P/L)	S_0 (mg P/kg)	BC (L/kg)	k (L/mg)	S_{max} (mg P/kg)	SC (mg P/kg)	% P Saturation
CN IA 1	VTA	1.25	73	**52**	0.092	**732**	**659**	10.1
	Grass	1.44	40	**26**	0.055	**556**	**516**	7.3
	p-value	0.631	0.077	**0.005**	0.140	**0.010**	**0.022**	0.268
CN IA 2	VTA	**0.00**	**0**	39	0.040	**997**	**997**	**0.0**
	Grass	**1.70**	**66**	34	0.071	**595**	**529**	**10.8**
	p-value	**<0.001**	**<0.001**	0.554	0.220	**<0.001**	**<0.001**	**<0.001**
NW IA 1	VTA	**3.82**	**127**	**28**	**0.053**	**761**	**634**	**16.7**
	Grass	**0.79**	**40**	**47**	**0.114**	**525**	**485**	**7.7**
	p-value	**<0.001**	**<0.001**	**0.036**	**0.017**	**<.0001**	**0.017**	**<0.001**
NW IA 2	VTA	**2.93**	74	**37**	**0.053**	**847**	**772**	8.6
	Grass	**0.71**	47	**56**	**0.135**	**517**	**470**	8.7
	p-value	**0.006**	0.153	**0.040**	**0.002**	**<0.0001**	**<0.001**	0.944
SW IA 1	VTA	0.61	**60**	**90**	**0.183**	632	**571**	**9.7**
	Grass	0.35	**13**	**36**	**0.052**	735	**722**	**1.8**
	p-value	0.510	**0.014**	**<0.001**	**<0.001**	0.121	**0.016**	**0.003**
SW IA 2	VTA	**2.15**	**75**	30	0.059	**658**	**583**	**11.0**
	Grass	**0.72**	**16**	24	0.057	**470**	**453**	**3.5**
	p-value	**<0.001**	**0.003**	0.518	0.935	**0.006**	**0.036**	**0.004**
SEM		0.27	13	6	0.017	33	43	1.8

The first parameter investigated was the equilibrium phosphorus concentration. This value represents the solution phosphorus concentration where sorption and desorption are equal. Values determined at these sites ranged from a low of 0.00 mg P/L for the VTA at Central Iowa 2 VTA soil to a high of 3.82 mg P/L for the Northwest Iowa 1 VTA soil. These values are similar to those reported by Sui and Thompson (2000) for biosolids amended soils in Iowa, but are generally lower than those found by Zhang et al. (2009) for surface soil horizons in vegetative treatment areas on New York farms that had received runoff effluent application for a comparable amount of time. At four of the sites equilibrium phosphorus concentrations were significantly different in the VTA soil than grassed area soil samples. These were CN IA 2, NW IA 1, NW IA 2, and SW IA 2. At all these sites, except Central Iowa 2, the equilibrium phosphorus concentration was significantly greater in the vegetative treatment area soil than in the grassed area soil, indicating increased risk of soluble phosphorus losses in drainage water and the possibility of soluble phosphorus transfer to runoff water. Overall, the results indicated that using vegetative treatment areas will on average increase soil equilibrium phosphorus concentrations (p-value = 0.0301). Similarly, the native sorbed phosphorus was also significantly higher in VTA soil than in the grassed area soil (p-value = 0.0036). All sites except Central Iowa 2 showed this trend with three of the sites having significantly higher native sorbed phosphorus levels than the grassed area at that site. These results were expected as all the VTAs, except Central Iowa 2, had received and retained large amounts of phosphorus over the previous five years based on Mehlich-3 phosphorus test results. Central Iowa 2 was unique in that phosphorus inputs to its treatment area were low due to the effective removal in the vegetative infiltration basin.

No consistent trend across the sites was seen for phosphorus buffering capcity (p = 0.1184). The site-by-site trend also show this inconsistency with two of the sites, Central Iowa 1 and Southwest Iowa 1, having significantly increased buffering capacities, and two, Northwest Iowa 1 and Northwest Iowa 2, having significantly decreased buffering capacities. In all cases the buffering capacities reported here are similar to those reported by Sui and Thompson (2000) for an Iowa Mollisol receiving applications of biosolids. This would seem to indicate that these soils had a history of high levels of phosphorus application which is supported by the relatively high Mehlich-3 soil test P levels (90 – 300 mg Mehlich-3 P/kg soil) present prior to use as vegetative treatment areas (Andersen et al., 2011).

No general effect on phosphorus sorption strength, i.e., binding energy, was seen (p = 0.9747). This was surprising as Sui and Thompson (2000), Holford et al. (1997), and Iyamuremye et al. (1996) had all reported significant decreases in phosphorus binding strength with manure application. In our study two of the sites, Northwest Iowa 1 and Northwest Iowa 2, exhibited this pattern of significant decreases in binding energy; however, Southwest Iowa 1 exhibited a significant increase in binding energy. The other three sites showed no significant change in binding energy. The inconclusive results, i.e., both increases and decreases in the soil's phosphorus binding energy, are similar to those reported by Laboski and Lamb (2004), whom also found that manure application could either increase or decrease binding energy.

In general the results showed a strong trend of increasing maximum phosphorus sorption capacity with use as a vegetative treatment area ($p < 0.0001$). Results from individual sites also indicated this trend of increasing maximum phosphorus sorption capacity with all sites except Southwest Iowa 1, which had no statistical difference in maximum sorption capacity, having significantly higher sorption capacities in the VTA soil than in the grassed area soil. Similarly, Laboski and Lamb (2004) reported increases in the phosphorus sorption capacity of a Nicollet soil treated with manure and found that greater increases in phosphorus sorption capacity occurred at higher manure application rates. In most cases the increase in maximum phosphorus sorption capacity were of greater magnitude than increases in native sorbed phosphorus, so statistical results for remaining sorption capacity were similar to those of the maximum sorption capacity. In general the VTA soils had significantly greater ($p < 0.001$) remaining sorption capacity than the soil samples from the grassed area. These results were unexpected as we had hypothesized that the high phosphorus loading rates these systems received would fill up the soil's existing sorption capacity. This result has important implications for projecting the phosphorus saturation life expectancy of these vegetative treatment systems, indicating that they may be able to fix phosphorus for greater lengths of time than originally anticipated (Baker et al., 2010); however, without knowing the mechanism of this rejuvenation in phosphorus sorption capacity further projections of phosphorus saturation life expectancy are also uncertain. Although these results of increasing phosphorus saturation life expectancy were unexpected, they are not unprecedented. Similar increases in phosphorus sorption life were seen for the Muskegon wastewater treatment system (Hu et al., 2006) in Michigan.

Results indicated that use as vegetative treatment areas significantly ($p = 0.0497$) increased the percent phosphorus saturation of the soil samples. This result held true at three sites which individually exhibited significant increases in percent phosphorus saturation, these were NW IA 1, SW IA 1, and SW IA 2. An increase in percent phosphorus saturation was also seen at Central Iowa 1; however, at this site the change wasn't significant. Central Iowa 2 showed a significant decrease in phosphorus saturation in the VTA as compared to the grass while Northwest Iowa 2 soil remained unchanged. At Northwest Iowa 2 this was likely caused by a large increase in the soil's phosphorus sorption capacity while at Central Iowa 2 this was likely due to losses of easily desorbed phosphorus from the soil.

CONCLUSIONS

Phosphorus retention in vegetative treatment areas is very dependent on the phosphorus sorption and desorption properties of the soil. Our laboratory studies indicate that use of soil as a vegetative treatment area is likely to increase the phosphorus concentration of the amended soil and of phosphorus in the soil solution (as evidenced by the increase in equilibrium phosphorus concentration). However, our results also suggested that continued application of feedlot runoff has the potential to significantly increase soil phosphorus sorption capacity (S_{max}), with most sites experiencing increases of between of 175 – 400 mg P/kg soil in sorption capacity when compared to the grassed area soil samples, presumable due to the large amounts of calcium applied with these wastewaters. Despite these increases in phosphorus sorption capacity most soils also showed significant increases in their percent saturation with phosphorus as well as in their soil solution equilibrium phosphorus concentrations. These changes could be important as research has indicated that losses of soluble phosphorus can increase rapidly when the percent phosphorus saturation of the soil reaches a change point (roughly 10-30%) while the soil solution equilibrium phosphorus concentration provides a direct estimate of concentrations of phosphorus leaching through the soil. These results indicate that while the phosphorus sink potential of the soil may increase, resulting in vegetative treatment areas that provide longer lives than originally anticipated, the level of environmental protection provided by the system is declining due to the greater potential for loss of soluble reactive phosphorus from the treatment system. Future work should seek to evaluate how this phosphorus is partitioning in the soil, (i.e., what pools are accumulating this phosphorus and the mechanisms responsible for its retention in the soil without increasing native sorbed phosphorus). Additionally, investigations into the mechanisms increasing soil phosphorus sorption capacity could be beneficial for siting similar waste treatment systems.

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Chapter 8. Vegetative Treatment System Impacts on Groundwater Quality

Abstract. *Increased environmental awareness has prompted the need for improved feedlot runoff control. Vegetative treatment systems (VTSs) provide a cost effective option that may enhance environmental security by protecting water quality. Vegetative treatment systems are typically designed on the basis of hydraulic performance, which may result in excess application of some nutrients, specifically nitrogen and phosphorus. Groundwater quality monitoring is required to determine the effect, if any, that VTSs have on groundwater. Shallow groundwater (2 – 10 m) quality beneath six VTSs in Iowa was monitored over a four year period. Monitoring wells were located up-gradient, within, and down-gradient of the VTSs. Groundwater samples were collected on a monthly basis and analyzed for ammoniacal-nitrogen, chloride, nitrate-nitrogen, and fecal coliform. A trend analysis was conducted to evaluate groundwater response patterns to VTS construction and use. In general, monitoring wells located within and down gradient of the VTS showed increasing trends in chloride and decreasing trends in nitrate concentrations. No trends for fecal coliform or ammoniacal-nitrogen were seen. Statistical analysis was performed to test for differences between up-gradient, within, and down-gradient monitoring wells. In general, no differences in ammoniacal-nitrogen concentration were seen. Fecal coliform concentrations were generally highest at the within VTS monitoring well, but no difference was found between up-gradient and down-gradient concentrations. Chloride concentrations were generally significantly higher within and down-gradient of the VTS when compared to the up-gradient well; nitrate concentrations were generally significantly lower at these locations. Overall, VTSs do not appear to be significantly degrading groundwater quality at these locations.*

Keywords. *Feedlot runoff, vegetative treatment systems, vegetative treatment areas, vegetative infiltration basins, groundwater monitoring, groundwater quality,*

INTRODUCTION

Open-lot animal feeding operation (AFO) runoff has been recognized as a potential pollutant to receiving waters because it contains nitrogen, phosphorus, organic matter, solids, and pathogens. The U.S. Environmental Protection Agency (EPA) developed a set of effluent limitation guidelines (ELGs) that described the design and operating criteria for feedlot runoff control systems on concentrated animal feeding operations (CAFOs) (Anschutz et al., 1979). These effluent limitation guidelines historically required collection, storage, and land application of feedlot runoff; however, recent modifications allowed the use of alternative treatment systems when the performance of the alternative systems, based on the mass of nutrients released, was equivalent to or exceeded that of an appropriately sized containment system (EPA, 2006).

Vegetative treatment systems (VTSs) are one possible alternative runoff control technology that has been proposed. A VTS is a combination of treatment components, at least one of which utilizes vegetation, to manage runoff from open lots (Moody et al., 2006). Vegetative treatment areas (VTAs) and vegetative infiltration basins (VIBs) are two possible treatment component options for VTSs. A sloped VTA is defined as an area level in one dimension, to facilitate sheet flow, with a slight slope along the other dimension, planted and managed to maintain a dense stand of vegetation (Moody et al., 2006). Operation of a VTA consists of applying solid settling basin effluent uniformly across the top of the vegetated treatment area and allowing the effluent to sheet-flow down the slope (Moody et al., 2006). Ikenbery and Mankin (2000) identified several possible methods in which effluent was treated by VTAs, including settling solids, infiltrating runoff, and filtering as it flowed through the vegetation. A VIB is defined as a flat area, surrounded by berms, planted to permanent vegetation (Moody et al., 2006). Effluent is distributed over the VIB surface via a flood effect. Vegetative infiltration basins have drainage tiles located 1 to 1.2 m (3.4 to 4 ft) below the soil surface to encourage infiltration of effluent. The tile lines collect effluent that percolates through the soil profile. The effluent then receives additional treatment, often from a VTA. Nutrient and pathogen removal in the VIB relies on effluent filtration as it percolates through the soil, plant uptake of nutrients, and microbial degradation of the nutrients and pathogens by soil fauna (Moody et al., 2006).

Two design approaches, one utilizing a hydraulic balance and the other a nitrogen balance, have been proposed for sizing VTAs (Woodbury et al., 2006). Previous work by Woodbury et al. (2005) has shown that if designed using the nitrogen balance approach, VTSs can successfully utilize applied nitrogen. However, in many cases VTSs have been designed based on hydraulic performance. This typically resulted in smaller VTSs, which may enhance opportunities for deep percolation of runoff water below the root zone and over-application, i.e., in excess of agronomic demand, of nutrients, especially nitrogen (Woodbury et al., 2006). As VTSs rely heavily on the soil-plant system to filter nutrients and contaminants before water percolates through the soil profile, there is a need to understand the impacts VTSs designed on a hydraulic performance have on groundwater quality. Thus far only limited data on the impacts of VTSs on groundwater quality is available, but work (Faulkner et al., 2011; Woodbury et al., 2005; Kim et al., 2006; and Schellinger and Clausen, 1992) has been inconclusive with Woodbury et al. (2005) detecting no percolation within the VTS they monitored and Faulkner et al. (2011) suggesting that groundwater impairment was unlikely based on their monitoring results, while Kim et al. (2006) and Schellinger and Clausen (1992) reported poorer treatment, i.e., higher concentration levels in leachate, which they thought was due to preferential flow.

The objective of this study was to evaluate the impact VTS installation and use had on groundwater quality. This manuscript reports results from a four-year groundwater monitoring study at six VTS locations on open beef feedlots in Iowa. A trend analysis was used to evaluate temporal patterns in the groundwater concentrations, specifically those of chloride, as it can serve as an indicator of manure transport, and nitrate, as it is of environmental concern. The results of the trend analysis were used to compare groundwater concentrations before-and-after VTS use. An analysis of variance was then used to compare groundwater concentrations up-gradient, within, and down-gradient of the VTSs.

MATERIAL AND METHODS

Site Descriptions

The performance of six vegetative treatment systems was monitored. These treatment systems were located on CAFO open beef feedlots throughout the state of Iowa. At many of the locations more than one VTS was installed. At these sites, one of the VTSs was monitored by Iowa State University (ISU) for nutrient release from each system component; only effluent released from the final treatment component of the other treatment systems (those not monitored by ISU) was performed. Table 1 shows the VTS configuration, the number of head, and the feedlot (55555+and additional drainage if present), VIB (where applicable), and VTA areas for both the on-farm and ISU monitored portions of the feedlot and runoff control system. Groundwater wells were sited and installed at each farm by an Iowa DNR geologist. Maps showing locations of the wells in relation to the feedlot and VTS location are shown in Figures 1a-f. Full descriptions of these sites were provided in Andersen et al. (2009); brief descriptions are provided here.

Central Iowa 1 (CN IA 1) was a 3.09 ha feedlot permitted for 1,000 head of cattle. Runoff effluent drained into a solid settling basin designed to hold 4,290 m^3 of effluent. A gate valve on the SSB outlet was used to control release volumes and rates onto the VTA. The VTA consisted of two channels operated in parallel; each channel was 24 m wide and averaged 311 m long. Central IA 1 VTA soil consisted of Clarion loam, Cylinder loam, and Wadena loam (Soil Survey Staff, NRCS USDA, 2010). Long-term average rainfall at this location was 85 cm of precipitation per year. Three groundwater wells were installed at Central Iowa 1 (CN IA 1). Depths for the up-gradient, within, and-down gradient were 7.8, 3.8, and 3.7 m, respectively; approximately the bottom meter of each well was screened. Average depths to groundwater were approximately 3.5, 0.65, and 1.1 m at the three locations, i.e., up-gradient, within, and down-gradient respectively.

The VTS at Central Iowa 2 consisted of a SSB, VIB, and VTA. Runoff from the 1.07 ha feedlot drained into a concrete SSB with a volume of 50 m^3. Prior to reaching the SSB outlet pipe, the effluent flowed through a "fence" of round bales. A gate valve controlled when, how much, and at what rate effluent was released .The outlet from the settling basin released effluent into the 0.32 ha VIB. Effluent from the VIB was pumped onto a 0.2 ha VTA. Soils in the VIB consisted of Nicollet loam and Webster clay loam and the VTA was Harps loam (Soil Survey Staff, NRCS USDA, 2010). Long-term average annual precipitation in this region averages 89 cm. Three groundwater wells were installed at Central Iowa 2 (CN IA 2). Well depths were approximately 4 m; approximately the bottom meter of each well was screened. Average depths to groundwater were approximately 1.5, 1.5, and 1.2 m at the up-gradient, within, and down-gradient locations.

Northwest Iowa 1 (NW IA 1) consisted of a 2.91 ha feedlot permitted to hold 1,400 head of cattle. Feedlot runoff was collected in a 1.2 m deep SSB having a volume of 3,700 m^3. The SSB outlet pipe discharged effluent uniformly along a concrete spreader located across the top width of the 1.68 ha VTA. A valve was used to actively control release of effluent from the SSB to the VTA. The Northwest IA 1 VTA soil consisted of Galva silty clay and Radford silt loam (Soil Survey Staff, NRCS USDA, 2010). Long-term average rainfall at this location was 66 cm of precipitation per year. Three groundwater wells were installed at Northwest Iowa 1 (NW IA 1); the wells installed were up-gradient, within VTS 1, and within VTS 2. Well depths for the up-gradient, within VTS 1, and within VTS 2 wells were 6, 9, and 6 m respectively. Approximately the bottom meter of each well was screened. Average depths to groundwater were approximately 3.7, 3.9, and 1.9 m at the three locations. Based on groundwater level monitoring it appeared that the general direction of groundwater flow was towards the within VTS 2 well from both the up-gradient well and within VTS 1 well.

Table 1. Description of whole farm and Iowa State University monitored vegetative treatment systems at the Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1(SW IA 1), and Southwest Iowa 2 (SW IA 2) feedlots. Information includes the number of head of cattle, the VTS configuration, and the size of the feedlot, settling basin (SSB), vegetative infiltration basin (VIB), and vegetative treatment area (VTA).

	# of Cattle		VTS Components		On Farm				ISU Monitored			
Site	Farm	ISU	On Farm	ISU Monitored	Feedlot (ha)	SSB (m^3)	VIB (ha)	VTA (ha)	Feedlot (ha)	SSB (m^3)	VIB (ha)	VTA (ha)
CN IA 1	1500	1000	2 SSB - 3 VTA	1 SSB - 2 VTA	4.11	5639	--	2.14	3.09	4289	--	1.49
CN IA 2	2400	650	3 SSB - 5 VIB - 2 VTA	1 SSB - 1 VIB - 1 VTA	3.26	136	1.09	0.72	1.07	51	0.32	0.22
NW IA 1	3400	1400	3 SSB - 5 VTA	1 SSB - 1 VTA	8.92	8906	--	4.06	2.91	3710	--	1.68
NW IA 2	4000	4000	1 SSB - 1 VIB - 1 VTA	1 SSB - 1 VIB - 1 VTA	2.95	110	1.01	0.60	2.96	110	1.01	0.60
SW IA 1	2300	2300	1 SSB - 10 VTA	1 SSB - 10 VTA	7.49	11550	--	4.05	7.49	11550	--	4.05
SW IA 2	5500	1200	12 SSB – 7 VTA	1 SSB - 1 VTA	19.67	33180	--	18.4	3.72	6275	--	3.44

Northwest Iowa 2 (NW IA 2) had an SSB-VIB-VTA system designed to control runoff from a 2.96 ha concrete feedlot. A settling basin with 1,120 m^3 capacity collected the feedlot runoff. Effluent from the settling basin was released onto a 1.01 ha VIB. The VIB had 15 cm diameter perforated tiles installed 1.2 m deep and spaced 4.6 m apart. Flow from the tile lines was collected in a sump and pumped onto the VTA. A gated pipe was used to spread flow evenly across the top width of the VTA. The 0.6 ha VTA was divided into two 27 m wide channels. At a given time, effluent was pumped onto only one of the VTA channels. The channel receiving effluent was switched manually by the producer. Northwest IA 2 consisted of Moody silty clay loam (Soil Survey Staff, NRCS USDA, 2010). Long-term average annual precipitation at this location was 66 cm of precipitation per year. Two groundwater wells were installed at Northwest Iowa 2 (NW IA 2). Well depths for the up-gradient and down-gradient wells were 9 and 6 m respectively. The bottom meter of the each well was screened. Average depths to groundwater depths at the up-gradient and down-gradient wells were 5.7 and 3.4 m respectively.

Southwest Iowa 1 (SW IA 1) was a 7.49 ha feedlot. Runoff drained into an 11,550 m^3 solid settling basin. A gate valve on the settling basin outlet was used to control SSB releases to the VTA. The 4.05 ha VTA was divided into ten channels. Effluent reaching the bottom of each VTA channel was then directed to the western most VTA channel. The VTA outlet was located 0.6 m above (in elevation) the bottom of the westernmost channel. This provided storage of effluent in the VTA before a release would occur. Tile lines, installed to control water table depth below the system and enhance infiltration of effluent into the soil, surrounded each of the VTA channels. The tiles were 1 m below the surface, 15 cm (6”) in diameter, and ran along the edges of each channel to a main running along the bottom of the VTAs. A tile access point was installed in early 2008 to monitor the amount and quality of flow in the tile lines. This point was located such that all flow was from the vegetative treatment area. Soils in the VTA consisted of mostly Judson silty clay loam and smaller areas of Colo-Ely complex (Soil Survey Staff, NRCS USDA, 2010). Long-term average annual precipitation in this area was 91.5 cm. Two wells were installed at Southwest Iowa 1 (SW IA 1). The locations of these two well do not allow for analysis of the impact of the VTS, but instead test the impact of the feedlot. Depths for both wells were approximately 6.1 m; average depths to groundwater were 1.9 and 2.9 m for the up-gradient and down-gradient wells respectively.

Southwest Iowa 2 (SW IA 2) was a 3.72 ha feedlot. Runoff drained into a solid settling basin designed to hold a 25-year, 24 h storm. A gate valve was installed on the settling basin outlet to control effluent release onto the VTA. The 3.44 ha VTA was constructed with earthen berm, level spreaders, along the length. The spreaders slowed the flow of effluent through the system, increasing the time for infiltration to occur. Southwest IA 2 VTA soil consisted of Kennebec silt loam (Soil Survey Staff, NRCS USDA, 2010). Long-term average rainfall at this location was 92 cm of precipitation per year. Three groundwater monitoring wells were installed at Southwest Iowa 2 (SW IA 2) to collect samples up-gradient, within, and down-gradient of the VTS. Average water table depths at the three well locations were 5.5, 2.4, and 5.1 m respectively. Groundwater depth monitoring indicated that the up-gradient well is truly up gradient; however, the “down-gradient” well is most likely operating as a second within VTS monitoring well and is not truly down-gradient of the VTS.

Monitoring Methods

Groundwater samples were collected monthly (between the 1st and 15th of the month) from each monitoring well and tested for ammoniacal-nitrogen (NH_3/NH_4-N), chloride (Cl^-), nitrate-nitrogen (NO_3-N), and fecal coliform (FC) concentrations. Occasionally wells were dry and no sample could be collected. Prior to sample collection stagnate water was purged from the well. The well was then allowed to recharge for five to seven days, after which a 250 mL sample was collected (100 mL of no treatment, 100 mL of acid treatment, and 50 mL in a sterile bottle for fecal coliform enumeration). After collection, the sample was placed on ice and shipped to a certified laboratory for analysis following chain-of-custody protocol. At the sites flow volumes from each of the treatment components, i.e., the settling basin, vegetative infiltration basins, and vegetative treatment area were monitored for flow volumes and nutrient concentrations on an event-by-event sampling basis to determine nutrient loading on each component (see Andersen et al., 2009 for sampling details).

Concentration Data Analysis

Regression analysis was used to analyze temporal trends in the groundwater concentration data, specifically those of chloride and nitrate. The regression equation fit was for an intervention at an unknown time. This equation fitted the data to three distinct phases. The first phase of the equation was a “stationary” mean, i.e., the average concentration before VTS construction and use. At the intervention point, the equation began a linear concentration increase or decrease phase, which occurred until the concentrations reached a new mean. The linear increase or decrease portion indicated how quickly the VTS was affecting groundwater concentrations. Where applicable, the third stage of the equation represented the average groundwater concentration after implementation and use had created a new approximately steady-state groundwater concentration. The equation used to model groundwater concentrations is shown in Eq. 1. In this equation C_i represents the sample concentration at the i^{th} sampling time, B_0 is average concentration before construction of the VTS, λ is the rate of change in groundwater concentration per day during the linear increase/decrease phase, τ_1 is the lag time (in days) before the linear increase/decrease phase begins, τ_2 is the lag time (in days) until the linear increase/decrease ends, $I_{[a,\infty)}$ is a step-function defined as 0 for all times less than *a* and as 1 for all times greater than or equal to *a*, ε_i is the model-fit error of the i^{th} sampling time, and t_i is a count variable that tracks the number of days since the background water sample was collected. Equation 1 was fit to the monitored data using the solver function in Microsoft Excel to minimize the sum of squares of error between the monitored and modeled concentrations.

$$C_i = B_0 + \lambda(t_i - \tau_1)I_{[\tau_1,\infty)} - \lambda(t_i - \tau_2)I_{[\tau_2,\infty)} + \varepsilon_i \qquad (1)$$

After fitting Eq. 1, a before-and-after analysis was performed for each well to determine if the change in average concentration was significant. This analysis was performed in Microsoft Excel as a comparison of means and assuming the variances for both time periods, i.e., before and after intervention, were homogeneous. Variances were estimated based on the residual error between the fitted model and the measured concentrations. An analysis of variance procedure was also used to test for differences between up-gradient, within VTS, and down-gradient wells. Only the concentration measurements falling into the third phase of Eq. 1 were used to evaluate differences between locations. The analysis was conducted as a REPEATED measures experiment using the PROC MIXED command in SAS 9.2. The analysis was conducted on a per site basis; location (i.e., up-gradient, within VTS, and down-gradient) was considered a fixed factor.

Estimating Leaching Volumes and Masses

Along with evaluating the trends in chloride and nitrate concentration in groundwater below the vegetative treatment area, estimating the mass of these parameters leached also provides significant insight into system performance and environmental impacts as it provides an assessment of potential nutrient loss mechanisms. The leached volume was estimated based on a cumulative water balance as shown in Eq. 2.

$$L = P + I - R - ET - \Delta S \qquad (2)$$

Where L is the volume of water leached (m^3/ha), P is the volume of water added through precipitation (m^3/ha), I is the volume of the water added via effluent application (m^3/ha), R is the volume of water lost via runoff from the VTA (m^3/ha), ET is volume of water evaporated and transpired from the VTA (m^3/ha), and ΔS is the change in soil moisture occurring during the monitoring period (m^3/ha). Precipitation depths were measured using an ISCO 674 tipping-bucket rain gauge (Teledyne ISCO, Lincoln, NE). A manual rain gauge installed on site was used to ensure rainfall data accuracy. Iowa Environmental Mesonet data (http://mesonet.agron.iastate.edu/) for the location closest to each site were used to determine precipitation depths for events occurring between 1 November and 1 April, mostly snowfall. Volumes of I and R were measured using ISCO 6712 sampler (Teledyne ISCO, Lincoln, NE) equipped with either ISCO 750 low-profile area-velocity sensor (Teledyne ISCO, Lincoln, NE) for pipe outlets or ISCO 720 submerged probe (Teledyne ISCO, Lincoln, NE.) in conjunction with a 0.45 m (1.5 ft) H-flume for non-pipe outlet locations. ET volumes were determined using the SPAW (Soil-Plant-Air-Water, Saxton et al., 2006) model to simulate the hydraulic budget of the site based on monitored site and weather conditions. Change in soil moisture, ΔS, was then assumed to be negligible in the water balance over the course of the multi-year monitoring period.

At sites with a within VTS well (Central Iowa 1, Central Iowa 2, Northwest Iowa 1, and Southwest Iowa 2) the estimated leached volume was multiplied by the monitored groundwater concentration from the within VTS well using the steady-state values as determined using Eq. 1. The groundwater sample was assumed to represent the concentration of the leachate as empirical evidence suggests that the large volumes of effluent applied in the vegetative treatment areas cause water table mounding (Machusick et al., 2011), i.e., the water table within the VTA would be higher than the surrounding landscape. Monitoring at these sites suggests this was occurring as water table levels within the VTA were typically higher than those monitored before system operation commenced. Additionally, results indicated that in many cases chloride levels monitored at the within VTA well have approached those of the applied effluent, indicating little mixing with groundwater flow at the within VTS well. No within VTS well was installed at Northwest Iowa 2, thus an alternative method was used to determine the mass leached at this site. In this case we used Eq. 3, which is based on a mass balance of chloride and assumes transport is dominated by convection, i.e., diffusion of chloride is negligible.

$$Q = L\frac{C_{down} - C_{ave}}{C_{up} - C_{down}} \tag{3}$$

This equation represents a mass balance of a conservative tracer, in this case chloride. *Q* represents the volume of groundwater flow through the upper end of the VTA (m^3), *L* the volume of leachate (m^3), C_{up} is the concentration of chloride in the up gradient well (mg/L), C_{ave} is the average concentration in the applied effluent (mg/L) corrected for plant uptake, precipitation, VTA release, evapotranspiration and scaled based on relationship between applied chloride concentration and within VTS concentration of the other sites. C_{down} is the concentration of chloride in the down gradient well (mg/L). Groundwater concentrations were taken as values obtained for the new steady-state conditions as determined in the trend analysis. The value of flow, *Q*, was then used in Eq. 4 to determine the concentration of nitrate-nitrogen in the leachate. This concentration was then multiplied by the leaching volume to determine nitrate-nitrogen leaching. This analysis relies on several assumptions, most notably, that vertical leakage of groundwater through the aquatard below the monitored water table is negligible and that transport is dominated by convection.

$$C_{leach} = \frac{(Q+L)C_{down} - QC_{up}}{L} \tag{4}$$

Due to the siting of the groundwater wells at Southwest Iowa 1 neither of these methodologies could be used; however, tiles were installed around the VTAs at this site. Flow and concentrations measurements from these tiles provide a measurement of the mass of chloride and nitrate leaching from the VTA, although additional leachate may not have been intercepted by the tiles.

Soil Sampling

Soil sampling was conducted prior to, and then again after approximately two and three years of, system operation. A soil sample was collected near the inlet and outlet of each ISU monitored VTA channel. During the initial soil sampling GPS coordinates were recorded for every sample location so the same spot would be sampled in subsequent years. This allowed change in soil nutrient content with time to be tracked at various positions in the VTA. At each soil sampling location, a soil sampling probe (Giddings Machine Company, CO) was used to collect a 2.54 cm (1 inch) diameter soil core that was 122 cm (48-inches) long. The sample was cut into segments to represent the 0-15.4 cm (0-6 inches), 15.4-30.5 cm (6-12 inches), 30.5-61 cm (12-24 inches), 61-94.4 cm (24-36 inches), and 94.4-122 cm (36-48 inches) depths. Each of these segments was put in a soil sampling bag and sent to the Soil and Plant Analysis Lab at Iowa State University for analysis of KCl extractable NO_3-N and NH_4-N. Average concentrations for both parameters were calculated for each depth increment;

the calculated average was assumed to represent the concentration of the midpoint of each sampling depth.

RESULTS AND DISCUSSION

Trend Analysis

The trend analysis was conducted by fitting Eq. 1 to the monitored concentration data for each parameter at each well and at each location. No trends for ammoniacal-nitrogen or fecal coliform concentrations were found for any well at any site. Trends in chloride and nitrate were seen at most locations and are discussed below.

Chloride

Chloride is present in large quantities in the feedlot runoff; flow-weighted average concentrations of chloride in the settling basin effluent were 234, 205, 596, 456, 232, and 500 mg/L for CN IA 1, CN IA 2, NW IA 1, NWIA 2, SW IA 1, and SW IA 2, respectively (concentrations for CN IA 2 and NW IA 2 are for the VIB effluent as this was applied to the VTA). Chloride is relatively un-reactive, i.e., it is not sorbed to soil and only small amounts are incorporated into biomass (it can account for between 0.2-2% of dry mass). As such it was treated as a conservative tracer when analyzing and interpreting groundwater data.

A plot of the chloride trends at the Central Iowa 1 monitoring wells is shown in Fig. 2a. In this figure, the x-axis represents the number of days the system was in operation; the 0 value represents the date the background sample at each well was collected. VTA operation then commenced within one month. At CN IA 1 chloride concentrations in the up gradient well remained constant over the 3.5 years of monitoring; concentrations at the in the VTS and down gradient wells both increased after VTS operation began. Statistical results indicated that chloride concentrations were significantly different ($p < 0.0001$) after use of the VTS as compared to pre-VTS conditions at both the within VTS and down-gradient wells. Model fitting results indicated that within VTS chloride concentrations increased by 124 mg/L while down-gradient concentrations increased by 15 mg/L. Concentrations within the VTS and down-gradient wells quickly reached a new steady-state concentration, presumably due to the shallow depth to groundwater at this site. Groundwater chloride concentrations within the VTS well stabilized at an average of 200 ± 30 mg/L. The graphed data shows a cyclical pattern; groundwater concentrations decrease in the winter and increase in the summer; this follows the effluent application pattern for the VTA. Also of note are the high chloride concentrations at the up-gradient well (273 ± 36 mg/L). This well was located at the edge of the feedlot; it appears that leaching of chloride from the pen surface has lead to elevated levels. Previous work by Olson et al. (2005) and Maule and Fonstad (2000) have shown that chloride concentrations in groundwater near the feedlot are often elevated as they reported concentrations ranging from 18 – 664 mg/L depending on feedlot age and site conditions.

Trends for chloride concentrations in Central Iowa 2 ground water (Figure 2b) indicated decreasing concentrations at the up gradient well, no change in the down gradient well, and increasing concentration at the within the VTS well. Concentration changes in the up-gradient and within VTS wells were significant with p-values less than 0.0001. The increase (27 mg/L) in the VTS groundwater well indicates that this VTS is infiltrating wastewater as intended. The increasing trend was much slower at this site than at Central Iowa 1. Although this site also had a shallow depth to groundwater, the well screen was installed in a clay layer with low permeability which slowed chloride transport to the well and limited percolation of applied effluent, similar to the phenomena noted by Faulkner et al. (2011) on their New York VTS site with a shallow soil. The decrease in chloride concentrations at the up-gradient well was unexpected; however, investigations of site conditions prior to VTS installation indicated that feedlot runoff pooled around this well location. Construction of the VTS lowered effluent and chloride application on this area, reducing chloride loading to the groundwater near this well.

Northwest Iowa 1 (Figure 2c) had a constant chloride concentration in the up gradient well and significant ($p < 0.0001$) increases in both within VTS wells; increases in chloride concentration were 210 and 451 mg/L at the within VTS 1 and VTS 2 wells, respectively. The lag time before VTS 1 chloride concentration started increasing was larger than the lag time for VTS well 2. This was probably due to the greater depth to the water table at VTS well 1, resulting in increased travel time before chloride in the applied settling basin effluent leached to groundwater. However, the concentration in VTS well 2 stabilized after the VTS well 1 concentration. Water table monitoring at this site indicated that groundwater was flowing from the VTS well 1 towards VTS well 2, thus concentration at VTS well 1 would need to stabilize before VTS well 2 as it is serving as a chloride input to groundwater near this well. Results at Northwest Iowa 2 were similar to those at Northwest Iowa 1. Up gradient concentrations were stable over the 3 ½ years of monitoring. The concentration increase in the down gradient well was again significant at the 0.0001 level with concentration increasing by 158 mg/L. The deeper water table at this site again delayed the time before the groundwater concentration began to respond.

Chloride concentrations in the Southwest Iowa 1 (Figure 2e) groundwater remained constant. This was due to the siting of the monitoring wells. Both wells are installed up gradient to the VTS, thus the monitoring wells did not allow the true impact of the VTS to be assessed. The chloride trends at Southwest Iowa 2 (Figure 2f) were different than at the other locations. The concentrations in the VTS and the down gradient wells both decreased significantly ($p < 0.0001$), by 64 and 15 mg/L, after initiating use of the VTS. The groundwater concentration decreases were presumably due to improved effluent distribution over the VTS. Previously, feedlot runoff at this site was allowed to pool in a grassed area below the feedlot. The VTS now spreads the applied settling basin effluent over the VTA, rather than allowing unsettled feedlot runoff to pool in the location where the groundwater wells were installed. Groundwater concentrations in the up-gradient well remained constant.

A correlation analysis was used to relate chloride concentrations monitored in the VTS well (except for NW IA 2 where the down gradient well was used as no in VTS well was available and SW IA 1 where tile flow chloride concentrations were used) to the flow-weighted average chloride concentration (corrected for losses of chloride in VTA release and in harvested vegetation and volumes of water from precipitation , VTA release, and evapotranspiration) in the effluent concentration applied to the VTA (Table 2). The correlation analysis indicated a strong relationship between the applied chloride concentration and the chloride concentration in the groundwater (Pearson’s $r = 0.91$). With the exception of Central Iowa 2, which had its well located in a clay layer, chloride concentrations averaged 85% of the applied chloride concentration. At CN IA 2 chloride concentrations at the groundwater well were only 28% of the applied concentration, we hypothesize that the clay layer restricted percolation and limited the impact of effluent application on groundwater quality at this location.

Table 2. Applied effluent chloride concentrations and within VTS well groundwater chloride concentrations at Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2).

	Applied Effluent Cl^- Concentration mg/L	Groundwater Cl^- Concentration mg/L
CN IA 1	223	200
CN IA 2	228	64
NW IA 1	634	576
NW IA 2†	430	235
SW IA 1‡	175	180
SW IA 2	525	437

† Groundwater concentration from the down-gradient well
‡ Groundwater sample represents effluent in the tile lines around the VTA.

Nitrate-Nitrogen

Nitrate-nitrogen (NO_3-N) trend analysis was conducted in a manner similar to that of chloride. In general NO_3-N nitrogen application on these sites exceeded crop uptake; however, studies with high strength wastewater have indicated that 50-80% of the applied N could be lost through denitrification and ammonia volatilization (Reed et al., 1998; Crites et al., 2000; Johns et al., 2009). Several soil factors could lead to high denitrification rates including high level of soil moisture, neutral to slightly alkaline soil pH, warm soil temperatures, and high availability of nitrate and organic carbon (Firestone, 1982). Many of these conditions were present in these VTAs including high levels of soil moisture and organic carbon availability. Thus, these conditions along with the trend of decreasing NO_3-N at the within VTS and down gradient monitoring wells may indicate that denitrification was serving as a sink for a significant portion of the applied nitrogen; alternatively, since effluent was surface applied and soils are at neutral to slightly alkaline reaction a large fraction of nitrogen may have been volatilized as ammonia.

Central Iowa1 (Figure 3a) within VTS and down gradient wells showed decreasing trends in NO_3-N concentration with time. Original NO_3-N concentrations were 216 and 70 mg/L at these locations, respectively; after the linear decreasing trends had reached a new steady-state, concentrations averaged 11 and 26 mg/L. This indicated that there was less NO_3-N leaching potential under the current land use as compared to previous conditions. During the summers of 2007 (around day 400) and 2008 (around day 800) within VTS groundwater NO_3-N concentration exhibited an annual peak; this trend was not noted in 2009. These peaks occurred in late summer and may indicate that drier soil conditions during these periods were facilitating greater nitrate production than vegetative uptake and denitrification were capable of utilizing, with the absence a peak in 2009 possibly being due to greater vegetative uptake as greater yields were obtained that year. Up gradient nitrate concentrations remained relatively constant; however, there was a period of abnormally high concentrations between day 700 and 800. This corresponded to construction of a hoop building near the groundwater well. The construction disturbed the soil in this area and possibly mobilized nitrogen that had accumulated within the soil profile. Groundwater NO_3-N concentrations returned to normal after this flush of nitrate. The trends observed in the soil nitrate concentrations (Figure 4a) complement those observed in the groundwater. Prior to system operation nitrate concentrations in the surface soil (top 15 cm) averaged approximately 5 mg NO_3-N/kg with increasing nitrate concentrations observed deeper in the profile (up to 15 mg NO3-N/kg at the 94-122 cm depth). Two years after commencing system operations nitrate concentrations in the surface soil (top 15 cm) were substantially higher averaging 20-25 mg/kg in 2008 and 2009 (Fig. 4a); however, these elevated nitrate concentrations were confined to the upper profile as at depths greater than 30 cm the soil nitrate concentration was less than under the previous land use (row crop agriculture), potentially indicate a reduced nitrate leaching potential.

Central Iowa 2 (Fig. 3b) also showed a trend of decreasing NO_3-N in the in VTS well. Interestingly, a decreasing trend in NO_3-N concentrations was also seen in the up gradient well; this corresponded with the decreasing trend in chloride seen in this well. This may indicate that installation and use of the VTS improved effluent handling over previous conditions at this site. No trend in NO_3-N concentration was seen at the down gradient well. In general nitrate-nitrogen concentrations were consistently low (< 10 mg NO_3-N/L) at this site, which is in stark contrast to observations at other sites. Again soil samples at this site (Fig. 4b) tended to corroborate the patterns observed in groundwater as nitrate concentrations deeper in the soil profile were again lower than under the previous land use.

Northwest Iowa 1 (Fig. 3c) showed no trend in up gradient NO_3-N concentration, increasing NO_3-N concentrations in the within VTS well 1, and a decreasing trend in VTS well 2. At this site, the VTAs contributing flow to VTS well 1 are located higher in elevation than VTS well 2 VTA. We observed that this lead to dryer VTA conditions. This could lead to more consistently aerated conditions that encourage nitrification and possibly limit denitrification opportunities. The within VTS 2 well was positioned below a VTA that was lower in elevation, had a shallower groundwater table, and stayed consistently wetter. The wetter condition could encourage high rates of denitrification and potentially limit nitrification. Denitrification appears to have occurred as a continuous decrease in NO_3-N concentration was monitored through the end of 2009 despite that fact that groundwater inflows of nitrate were probably increases (as within VTS well 1 was up-gradient of within VTS well 2). These differing responses at with VTA wells 1 and 2 support a nitrification-denitrification treatment mechanism rather than volatilization of ammonia as the mechanism of nitrogen removal since ammonia volatilization would presumably resulted in lower nitrate concentrations at both groundwater wells; however, measurements of gaseous nitrogen emissions would be required to verify this hypothesis. Monitoring of soil nitrate concentrations at this site showed a similar pattern to those observed at CN IA 1, i.e., high nitrate concentrations in the surface soil but low concentrations deeper in the profile (Figure 4c). However, nitrate concentrations in the surface soil at this site where much higher (average values of 131 and 70 mg NO_3-N/kg in 2008 and 2009) than was observed in 2006 (15 mg NO_3-N/kg) or those observed at Central Iowa 1.

Northwest Iowa 2 (Fig. 3d) groundwater trends were similar to those observed at the NW IA 1 up-gradient and within VTA well 2 locations, i.e., nitrate concentrations remained constant in the up gradient well while the down gradient well showed a consistent trend of decreasing NO_3-N concentrations. At the down gradient well, NO_3-N levels were initially monitored to be 164 mg/L, by the end of the 3 ½ years of monitoring nitrate-nitrogen concentrations had stabilized at 15 mg/L. This would again indicate that the VTA exhibited reduced nitrate leaching potential than the previous land use (row crop production), and that the VTA is potentially encouraging denitrification as nitrate concentration has decreased below those monitored in the up-gradient well. The trends in soil nitrate concentrations (Fig. 4d) shared some similarities to those observed at other sites, but in this case elevated nitrate concentrations (> 80 mg NO_3-N/kg) down to a depth of 60 cm was observed in 2009. This may indicate the potential of nitrogen movement through the soil profile; however, below this depth concentrations dropped rapidly to average values lower than observed for the previous land use condition.

No trends in NO_3-N groundwater concentrations were seen at Southwest Iowa 1 (Figure 3e). This was again attributed to the monitoring well siting being around the feedlot and not the treatment system. Soil nitrate concentrations (Fig. 4e) exhibited a small decrease in nitrate nitrogen concentrations throughout the soil profile, but in both cases (before and after system use) the actual nitrate concentrations observed were relatively low, averaging between 2 and 8 mg NO_3-N/kg. Southwest Iowa 2 showed a small, but signification ($p < 0.001$) increase in NO_3-N concentration at the up gradient well. Model fits were extremely poor for the in VTS and down gradient monitoring wells; as such the models are not shown (Figure 3f). These two wells exhibited a sinusoidal pattern with maximum NO_3-N concentrations occurring during the summer and minimums occurring in the winter, similar to seasonal pattern noted at CN IA 1. This would seem to indicate that during the warmer, and drier, summer months larger amounts of the applied ammonium and organic nitrogen were being nitrified, increasing leaching potential. During the winter and spring nitrate-nitrogen concentrations would drop to levels near the detection limit. Groundwater level monitoring at this site indicated the presence of a seasonal high water table that lead to saturation of the soil profile in the winter and spring; during the summer the shallow water table dropped rapidly. These conditions wet conditions in the winter and spring could denitrification, but the higher oxygen availability during the summer and fall would favor nitrification. Soil sampling (Figure 4f) showed a markedly different response at this site than that observed at the other five sites as the deeper soil profile exhibited small increases in nitrate concentration (~1-2 mg NO_3-N/kg) as opposed to the decreases in nitrate observed at the other locations. However, similar to the other sites, the surface soil again had increased nitrate-nitrogen contents when compared to before system operation.

Effect of VTS on Groundwater Quality

Ammoniacal-Nitrogen

Most of the groundwater samples collected were at or below the ammoniacal-nitrogen detection limit of 0.20 mg NH_3N/L. When calculating statistics all samples that were reported as below detection limit were assumed to be at the detection limit. Averages, standard deviations, and significant differences are shown in Table 2. The majority (> 90%) of samples at CN IA 1, NW IA 2, and SW IA 1 were at or below detection limit (0.20 mg/L). At NW IA 1, more than 80% of samples were below the detection limit and no significant difference in ammoniacal-nitrogen concentrations was detected. At SW IA 2 ammoniacal-nitrogen was not detected at the up gradient well; the in VTS and down gradient well were significantly different from the up gradient well. The higher levels were present at the start of the study and may indicate previous contamination of shallow groundwater. The VTS area had received runoff from the feedlot for more than 30 years, thus the higher levels can probably be attributed to historic ammonium accumulation in the soil with the more even distribution of effluent over the VTA actually reducing the risk of detectable levels of ammoniacal nitrogen. At CN IA 2 ammoniacal-nitrogen was rarely, ~20% of the time, detected at the in the VTS monitoring well. Monitoring wells up-gradient and down-gradient of the wells were above the detection limit for more than 95% of the time. At this site all wells were significantly different from each other with the up-gradient well having the highest concentrations; the VTS well was the lowest.

Overall the groundwater monitoring results would seem to indicate that the use of VTSs was not causing ammoniacal-nitrogen concentrations in groundwater to increase; however, the deep soil sampling conducted within the VTA provides significantly more insight into what may be occurring. At many of the sites soil ammonium-nitrogen concentrations were elevated in comparison to under the previous land use. In many ways this result is unsurprising as they are dosed frequently with nitrogen, specifically ammonia, rich wastewaters. However, at two locations, CN IA 1 and NW IA 2 increases (~ 15 and 28 mg NH_4-N/kg respectively) in soil ammonium content were observed deeper (below 0.6 m) in the soil profile. At both sites this trend was only observed on cores collected near the VTA inlet. Moreover, these cores tended to have high nitrate contents in the surface and then low concentrations lower in the profile, i.e., opposite the trend observed for ammonia. This may indicate that different depths of the soil were experiencing different levels of aeration. The soil surface has dried and nitrogen in this soil depth has mineralized, while deeper in the soil profile the soil is still under anaerobic conditions.

Table 3. Average (standard deviation) ammoniacal-nitrogen concentrations in up-gradient, in vegetative treatment system (VTS), and down gradient monitoring wells for Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1(SW IA 1), and Southwest Iowa 2 (SW IA 2). Lower case letters represent significant differences at the α = 0.05 level within a row.

Site	Up Gradient	In VTS	Down Gradient
CN IA 1	0.22 (0.07)[a]	0.21 (0.07)[a]	0.20 (0.00)[a]
CN IA 2	3.02 (1.02)[a]	0.23 (0.07)[b]	1.37 (0.69)[c]
NW IA 1	0.36 (0.82)[a]	0.69 (2.30)[a]	0.51 (0.93)[a]
NW IA 2	0.20 (0.00)[a]	NA	0.21 (0.06)[a]
SW IA 1	0.21 (0.02)[a]	NA	0.21 (0.03)[a]
SW IA 2	0.20 (0.00)[a]	1.08 (2.05)[b]	0.65 (0.56)[b]

SW IA 2 exhibited the opposite trend in ammonium-nitrogen concentrations as seen at CN IA 1 and NW IA 2. As discussed previously, the in VTS well at SW IA 2 had numerous groundwater samples that were above the ammoniacal-nitrogen detection limit but this appeared to be due to background contamination. Soil samples appear to confirm this as high levels (200-250 mg NH_4-N) were observed at the lower depths in the soil profile during collection of the background soil sample. After several years of VTS operation these levels had decreased below 50 mg NH_4-N/kg and appear to continue to be decreasing.

Chloride

In general, it appeared that chloride concentrations in the VTS wells were higher than the up-gradient wells. The applied wastewater had high concentrations of chloride; chloride is relatively un-reactive, as such it can be used as a tracer of where manure or wastewater has been applied and infiltrated. The high chloride concentrations could be taken as an indication that the VTSs were infiltrating a large portion of the applied effluent. At CN IA 1 chloride concentrations at the up-gradient well were significantly higher than the in VTS and down gradient wells; this well was located near the feedlot and high concentrations are probably a direct result (see Olson et al., 2005 and Maule and Fonstad, 2000 for discussion of groundwater contamination around feedlots). At CN IA 2, chloride was highest at the within VTS monitoring well; concentrations in up-gradient and down-gradient wells were significantly different, but actual chloride concentrations were similar at 15 and 12 mg/L respectively. NW IA 1 and NW IA 2 both experienced chloride concentration increases at the in VTS and/or down-gradient wells as compared to the up-gradient well. Chloride concentrations at SW IA 2 were significantly higher at the in VTS well; the down-gradient well was also significantly higher than the up-gradient again indicating infiltration of wastewater. At SW IA 1 chloride concentrations at the down-gradient well were significantly lower than the up gradient well.

Table 4. Average (standard deviation) chloride concentrations in up-gradient, in vegetative treatment system, and down gradient monitoring wells for Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1(SW IA 1), and Southwest Iowa 2 (SW IA 2). Lower case letters represent significant differences at the α = 0.05 level within a row.

Site	Up Gradient	In VTS	Down Gradient
CN IA 1	273 (36)[a]	200 (30)[b]	71.0 (7.0)[c]
CN IA 2	15.4 (2.9)[a]	64.0 (7.1)[b]	12.1 (5.2)[c]
NW IA 1	54.4 (9.8)[a]	256 (82)[b]	576 (31)[c]
NW IA 2	55.8 (6.4)[a]	NA	235 (13)[b]
SW IA 1	48.1 (24.7)[a]	NA	17.7 (1.7)[b]
SW IA 2	8.14 (1.26)[a]	437 (34)[b]	63.2 (13.9)[c]

Nitrate-Nitrogen

Nitrate-nitrogen concentration differences seem to be very site specific as large differences occurred at different sites. At most of the sites (except CN IA 2) nitrate-nitrogen concentrations exceeded the

10 mg NO_3-N/L drinking water concentration, most notably at CN IA 1 the average concentration was 117 mg/L. As mentioned this well was located near the feedlot and it may be impacting monitored groundwater concentrations. Similarly Maule and Fonstad (2000) reported concentrations ranging from 2.5 – 233 mg NO_3-N/L. At CN IA 2, the down gradient well had a significantly higher average concentration than either the up-gradient or the VTS wells; however, actual concentrations were relatively low with an average of 2.52 mg/L. NW IA 1 NO_3-N concentrations were lowest at in VTS 2 well (shown as down gradient in Table 5) and highest at in VTS 1. As discussed previously, this well was sited at a location with a deeper depth to groundwater than in VTS 2. As a result, conditions in the VTS were drier and presumably facilitated nitrification and limited denitrification. The VTS well 2 was located below another VTA, that was lower in elevation and stayed much wetter; presumably facilitating denitrification as NO_3-N concentrations were reduced at this location. Similarly down-gradient NO_3-N concentrations at NW IA 2 were significantly lower than up-gradient concentrations. At SW IA 2 concentrations in the VTS were significantly higher than either up-gradient or down-gradient. Both the in VTS and down-gradient wells had high amounts of variability in NO_3-N concentrations; this was caused by the seasonal trend of higher NO_3-N concentration in summer and low concentration in winter. Overall it appears that VTSs are not causing significant increases in groundwater NO_3-N concentrations, and in some cases they are even reducing NO_3-N levels; however, seasonal trends of high NO_3-N concentrations in the summer were seen at several locations. More research to determine mechanisms that cause these trends in NO_3-N concentrations and to determine which sites would experience NO_3-N removal within the VTS is required.

Table 5. Average (standard deviation) nitrate-nitrogen concentrations in up-gradient, in vegetative treatment system (VTS), and down gradient monitoring wells at Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1(SW IA 1), and Southwest Iowa 2 (SW IA 2). Lower case letters represent significant differences at the α = 0.05 level within a row.

Site	Up Gradient	In VTS	Down Gradient
CN IA 1	117 (53)^a	11.0 (23.1)^b	26.0 (12.4)^b
CN IA 2	0.18 (2.03)^a	0.33 (0.19)^a	2.52 (2.74)^b
NW IA 1	19.0 (11.3)^a	57.7 (16.6)^b	3.80 (5.56)^c
NW IA 2	40.3 (5.1)^a	NA	15.3 (8.5)^b
SW IA 1	39.6 (12.5)^a	NA	0.18 (10.51)^b
SW IA 2	11.4 (1.2)^a	33.6 (32.1)^b	2.72 (19.4)^a

Fecal Coliform

The log value of all fecal coliform concentrations was taken. Statistical analysis was performed on the log values of the fecal coliform concentrations. With the exception of Central Iowa 2, fecal coliform concentrations were highest at the within VTS well. At Central Iowa 2 the monitoring well was installed in a clay layer that slowed percolation and reduced transport of contaminants to groundwater, similar to the function of the fragipan described at Faulkner et al. (2011) at their New York VTS site. At most sites (CN IA 1, CN IA 2, NW IA 1, NW IA 2, and SW IA 1) concentrations at the up- and down-gradient wells were not significantly different. At NW IA 1, concentrations at VTS 1 were significantly greater than the up gradient well. In VTS well 2 (shown as down gradient in Table 6) was not significantly different from either the up-gradient well or VTS 1. No impact was seen at the NW IA 2 or SW IA 1 monitoring wells. All three wells at SW IA 2 were significantly different from each other with concentration being highest at the in VTS well and lowest at the up-gradient well.

Table 6. Log values of average (standard deviation) fecal coliform concentrations in up-gradient, in vegetative treatment system (VTS), and down gradient monitoring wells at Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1(SW IA 1), and Southwest Iowa 2 (SW IA 2). Lower case letters represent significant differences at the α = 0.05 level within a row.

Site	Up Gradient	In VTS	Down Gradient
CN IA 1	2.24 (1.37)^{a}	2.63 (1.06)^{a}	1.55 (0.69)^{b}
CN IA 2	1.66 (0.84)^{a}	1.21 (0.49)^{b}	1.87 (0.89)^{a}
NW IA 1	1.28 (0.44)^{a}	1.71 (0.65)^{b}	1.60 (0.74)^{ab}
NW IA 2	1.30 (0.67)^{a}	NA	1.47 (0.69)^{a}
SW IA 1	1.47 (0.79)^{a}	NA	1.93 (1.09)^{a}
SW IA 2	1.49 (0.56)^{a}	3.70 (1.59)^{b}	2.17 (0.94)^{c}

Chloride and Nitrate-Nitrogen Leaching

Using the methods described previously the average volume of water and mass of chloride and nitrate leached was calculated. The calculation was based on a cumulative water balance to determine the amount of water potentially leached and the monitored concentration chloride and nitrate concentrations at the within VTS groundwater well. In general, these chloride leaching masses were 30-85% of the applied chloride masses with another 5-20% being removed with harvested vegetation. Thus, approximately 60-90% of the applied chloride can be tracked at these sites. Although far from perfect, this level of tracing provides strong evidence that the leaching estimates are reasonable. Following the same methodology NO_3-N leaching was estimated to range from 2-39 kg NO_3-N/ha-yr. At SW IA 2, where tile lines surrounded the VTA, approximately 28 kg NO_3-N/ha-yr was monitored in tile flow. This estimate is reasonable in comparison to the estimated leached masses of nitrogen occurring at the other five sites. In general, these results are similar to those of tile drained fields under a corn-soybean rotation in the upper Midwest. For instance, results from the Midwest have ranged from 0 – 50 kg NO_3-N/ha (Randall et al., 1997; Randall et al., 2003; Randall and Vetsch, 2005), while the work of Bahksh et al. (2005, 2006) found losses in Iowa were 11 to 14 kg N/ha. These nitrogen leaching losses only account for a small portion, i.e., 0.5-2%, of the applied nitrogen at these sites. This analysis, in conjunction with the measured groundwater nitrate-nitrogen concentration data, would seem to indicate that despite the high hydraulic and nitrogen loading these vegetative treatment areas are receiving; they are not causing excessive harm to groundwater resources.

Table 7. Volume and mass of chloride and nitrate-nitrogen estimated to be leached by the vegetative treatment areas at Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2) based on long-term hydraulic balances and monitored groundwater concentrations.

Site	Leached Volume (m^3/ha-yr)	Chloride (kg Cl^-/ha-yr)	Nitrate-Nitrogen (kg NO_3-N/ha-yr)
CN IA 1	3,600	710	39
CN IA 2	6,200	400	2
NW IA 1	5,800	3,300	22
NW IA 2	6,700	1,600	15
SW IA 1†	1,300	250	14
SW IA 2	4,200	1,800	11

† Leaving volume and masses estimate based on monitored tile flow measurements.

CONCLUSIONS

A trend analysis was conducted to evaluate groundwater chloride and nitrate response patterns to VTS construction and use. In general, monitoring wells located within and down gradient of the VTS showed increasing trends in chloride and decreasing trends in nitrate concentrations. No trends for fecal coliform or ammoniacal-nitrogen were seen. Statistical analysis was performed to test for differences between up-gradient, within, and down gradient monitoring wells. In general, no differences in ammoniacal-nitrogen concentration were seen with most samples being below the ammonia-nitrogen detection limit. Fecal coliform concentrations were generally highest within the VTS monitoring well but showed no difference between up-gradient and down-gradient concentrations. Chloride concentrations were generally significantly higher within and down-gradient of the VTS when compared to the up-gradient well; nitrate concentrations were generally significantly lower at these locations. Overall, it appeared that VTSs do not appear to be significantly degrading water quality at these locations. A water-balance model was then used to estimate volumes of water that were leached, which was used to estimate chloride and nitrate leaching. In generally, results suggested that 30-85% of the applied chloride was in the leachate; however, only 0.5-2% of the applied nitrogen was leached. Nitrate-nitrogen leaching masses were estimated to range from 2-40 kg/ha; these values are similar to those reported for corn-soybean rotation tile drainage in Iowa and suggest more study is needed to better understand the fate of the applied nitrogen.

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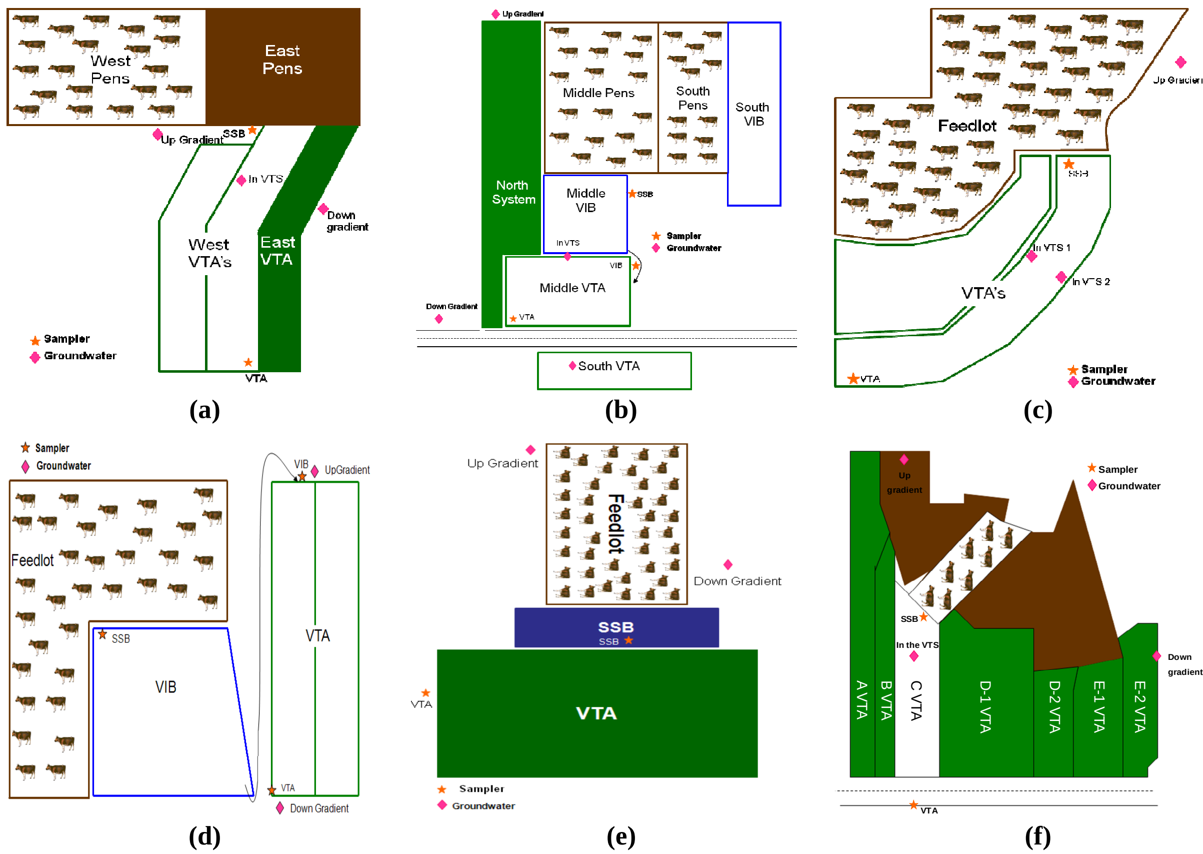


Figure 1. Groundwater well locations in relation to the feedlot and VTS components for (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2.

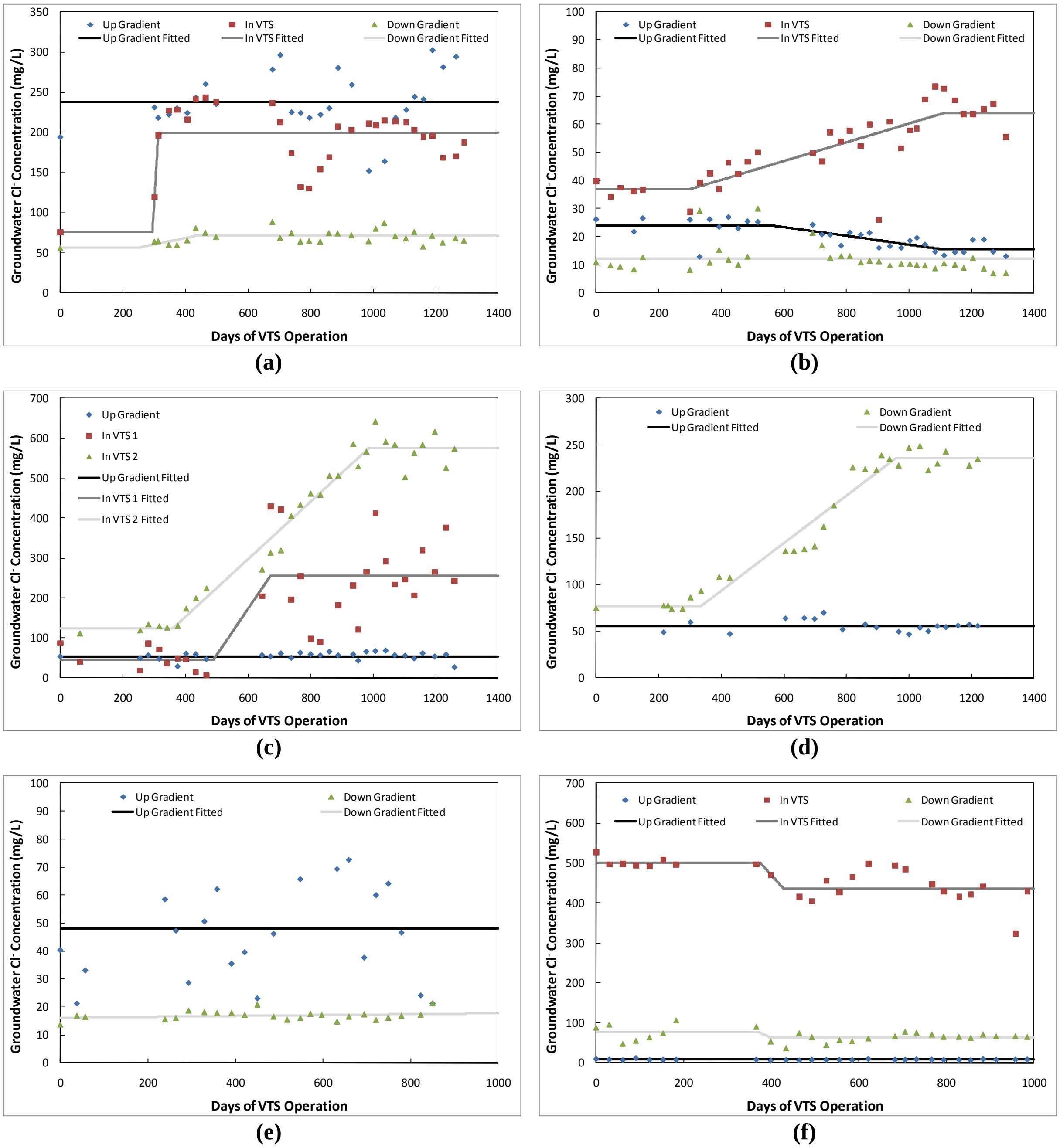


Figure 2. Groundwater chloride concentration trends at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

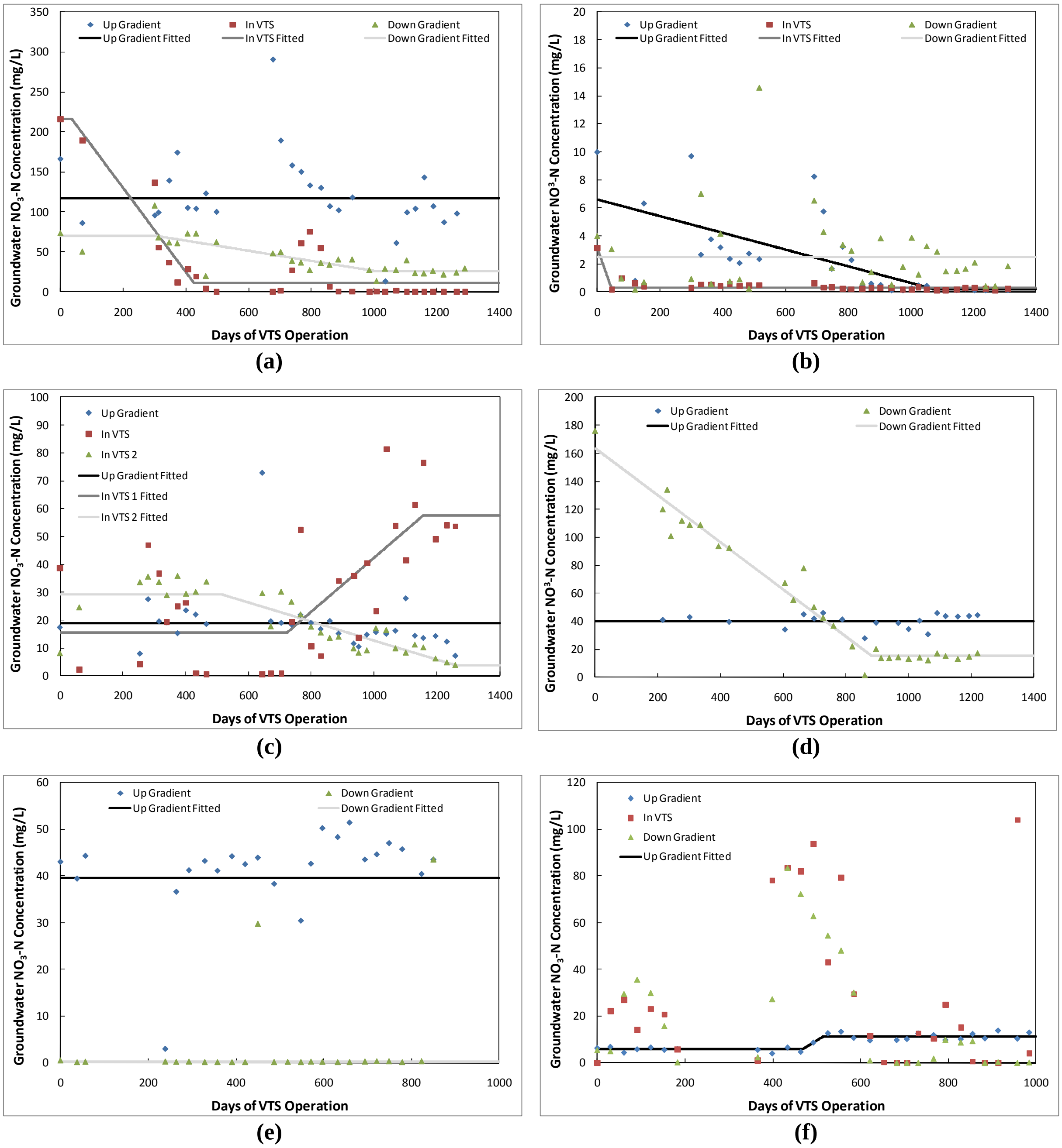


Figure 3. Groundwater nitrate-nitrogen concentration trends at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

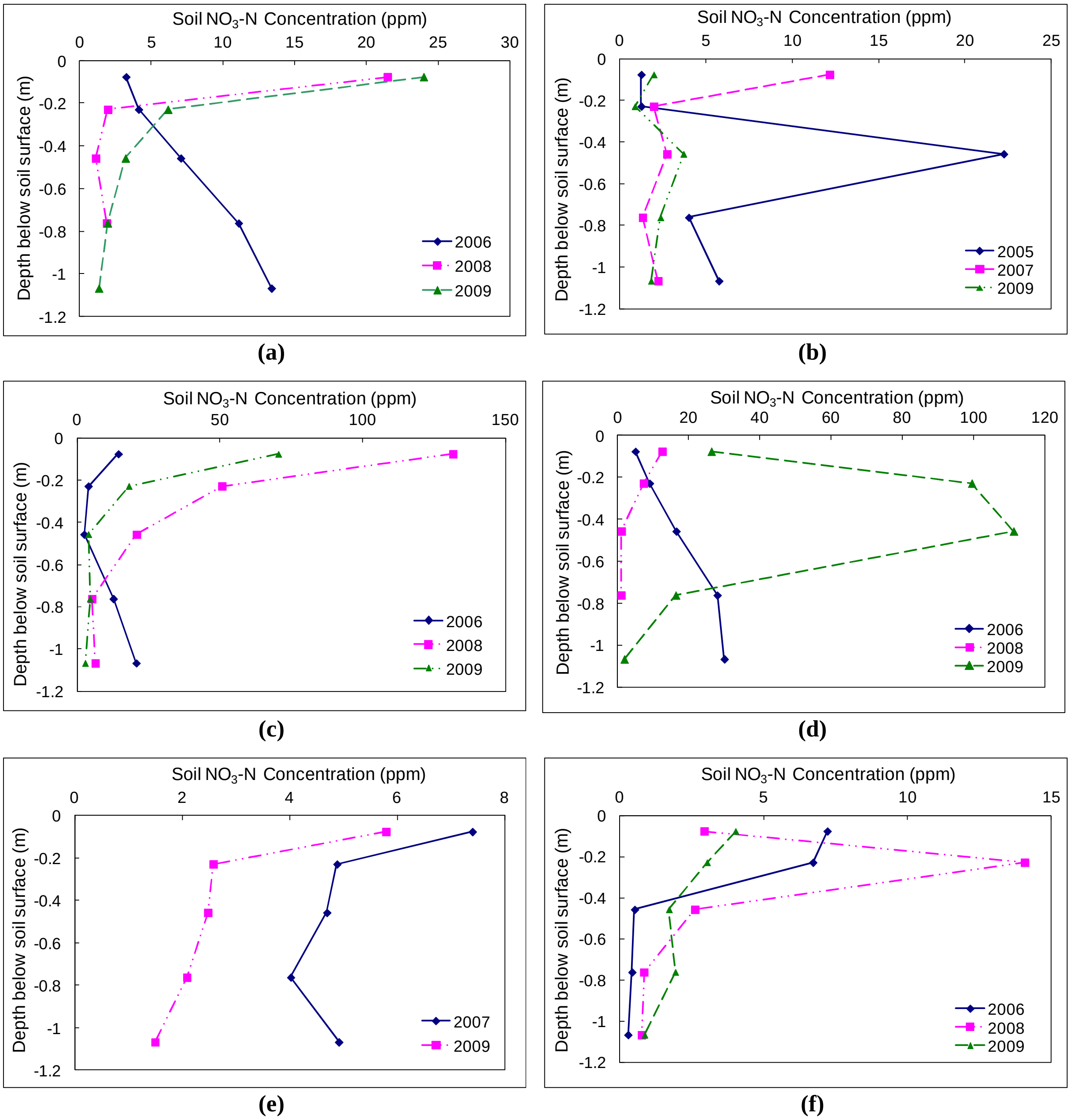


Figure 4. Soil nitrate-nitrogen concentrations as a function of depth at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

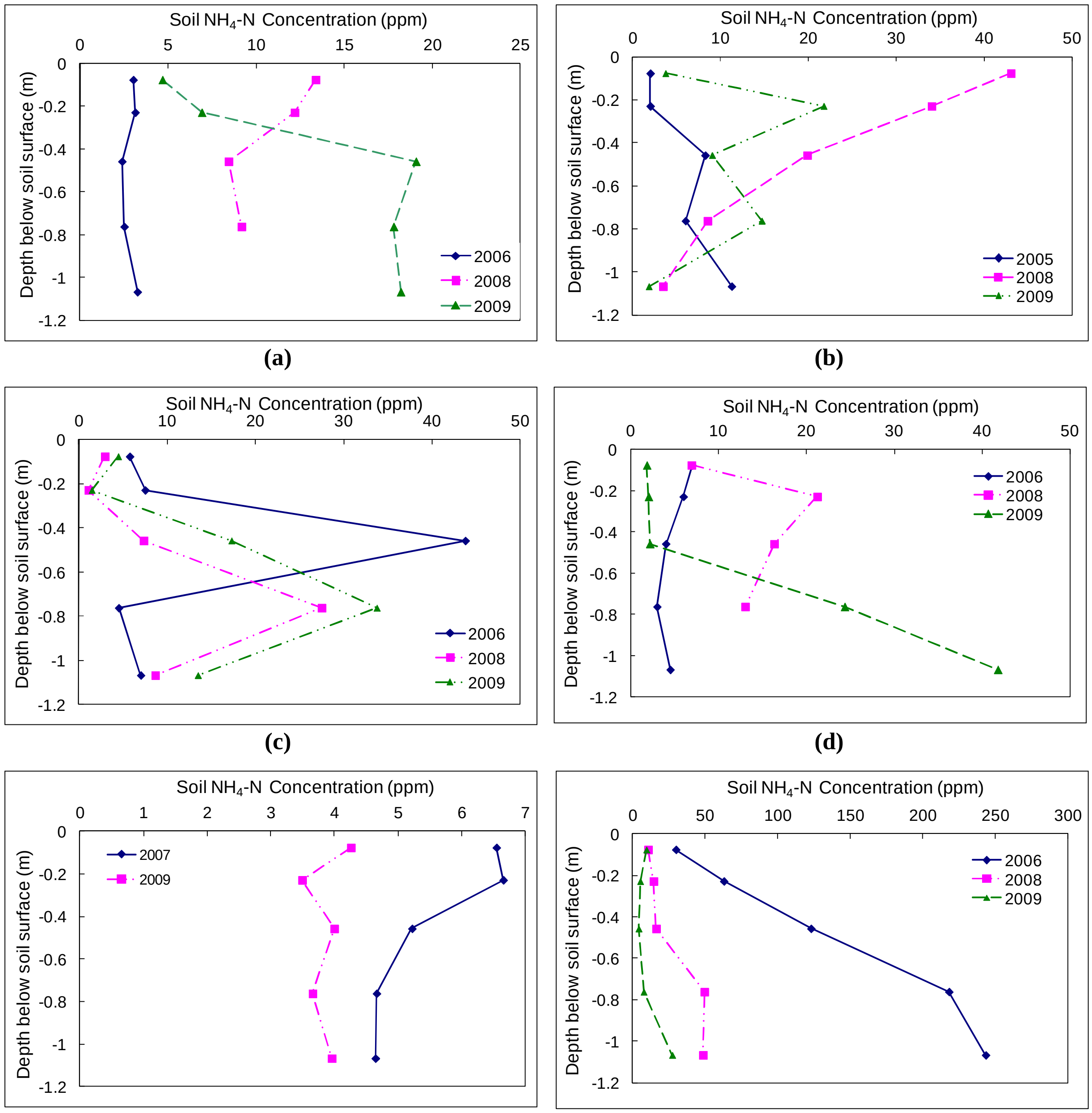


Figure 5. Soil ammonium-nitrogen concentrations as a function of depth at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

Chapter 9. The Impact of Vegetative Treatment Area Use on Soil Biologically Available Carbon and Nitrogen Pools

Abstract. *Vegetative treatment systems are being utilized to control and treat runoff from beef feedlots. These systems are grassed areas that rely heavily on the soil-plant system to control and treat oxygen demanding materials, nitrogen, and phosphorus present in runoff from open beef feedlots. In the short term they have proved effective and are playing a critical role in abating the point source pollution potential of the feedlot, but due to their nature as part of a waste disposal system they are subject to high nutrient loading, especially nitrogen. As such there is great interest in the impact this application has on the soil organic matter and on the ability of the soil to treat future feedlot runoff applications. Of specific concern are symptoms of nitrogen saturation (as seen in forest ecosystems subject to high levels of nitrogen deposition) as these are associated with reduced nitrogen retention within the ecosystem and increases nitrogen losses. One of the primary symptoms cited is nitrogen enriched of soil organic matter, as this is often linked with losses of oxidized nitrogen . Unfortunately, these changes are often hard to detect due the large size of the soil organic matter pool in comparison to the change in nutrient content. However, the labile component of soil organic matter can serve as an early indicator of these changes. We performed a long-term biological fractionation of soil from a vegetative treatment area (after five years of use) and from a paired grass area to evaluate if use of the vegetative treatment area to control and treat feedlot runoff has caused increases in soil labile carbon, labile nitrogen, or nitrogen enrichment of the labile fraction. Results indicated that while use of the soil as a vegetative treatment system has often increased the soil's labile carbon content, labile nitrogen increases were usually larger. This resulted in enrichment of nitrogen in the labile organic matter pools, indicating looser nitrogen cycling, which could make the vegetative treatment system more prone to nitrogen loss via leaching or gaseous emissions.*

Keywords. *Biological fractionation, labile carbon, labile nitrogen, soil carbon-to-nitrogen ratio*

INTRODUCTION

Vegetative treatment systems are being utilized to control and treat runoff from beef feedlots. A VTS is a combination of treatment components, at least one of which utilizes vegetation, to manage runoff from open lots (Koelsch et al., 2006). Vegetative treatment areas (VTAs) and vegetative infiltration basins (VIBs) are two possible treatment components for VTSs. A vegetative treatment area is a band of planted or indigenous vegetation situated down-slope of cropland or an animal production facility that provides localized erosion protection and contaminant reduction (Koelsch et al., 2006). A sloped VTA is an area level in one dimension, to facilitate sheet flow, with a slight slope along the other, planted and managed to maintain a dense stand of perennial vegetation (Moody et al., 2006).

Operation of a sloped VTA consists of applying solid settling basin effluent uniformly across the top of the vegetated treatment area and allowing the effluent to sheet-flow down the slope. Ikenberry and Mankin (2000) identified several possible methods in which effluent was treated by VTAs, including settling solids, infiltrating the runoff, and filtering of the effluent as it flowed through the vegetation.

These system have proven effective in the short term (<5 years), but as these systems rely heavily on the soil-plant system to control and treat oxygen demanding materials, nitrogen, and phosphorus present in runoff from open beef feedlots there are questions about the long-term sustainability of their treatment mechanisms. Specifically, there is concern that the soil organic matter will become nitrogen enriched and exhibit nitrogen saturation systems typical of east coast United States forests experiencing high levels of nitrogen deposition. Unfortunately, changes in soil organic matter are often small in comparison the size of the soil organic matter pool, making detection difficult. However, the labile component of soil organic matter plays an important role in short-term nutrient turnover (Tisdale and Oades, 1982) and is often more sensitive to management changes that total carbon or nitrogen, as such, it has been suggested that it could serve as an early indicator of future trends for the soil organic matter (Bremer et al., 1994). Thus, studying this pool can provide information on impacts management changes are having on soil quality in a timelier manner.

The objective of this study was to perform a long-term (>1 year) biological fractionation of soil from six vegetative treatment areas and paired soils from grasslands at each site to evaluate if increases in labile carbon and nitrogen had occurred. These data were also utilized to evaluate if the labile pool was becoming nitrogen enriched, i.e., if it was exhibiting signs of nitrogen saturation.

METHODS AND MATERIALS

Site Descriptions

Six vegetative treatment systems were located on concentrated animal feeding operation (CAFO) sized open beef feedlots throughout the state of Iowa and intensively monitored over a four year period by Iowa State University. The sites were described in detail in Andersen et al. (2009) and are only briefly discussed here. Data summarizing the characteristics of the Iowa State University (ISU) monitored portions of the feedlots and VTSs are provided in table 1. Information shown includes the maximum cattle capacity of the feedlot, the VTS configuration, the size of the drainage area (feedlot and additional contributing area), the volume of the settling basin, the area of the VIB (where applicable), and the area of the VTA. Characteristics of the sites are discussed below.

Table 1. Summary of the system configuration and vegetative treatment system components at each site.

Site	No. of Cattle	VTS Components	Drainage Area (ha)	SSB (m^3)	VIB (ha)	VTA (ha)
Central Iowa 1	1,000	1 SSB - 2 VTA	3.09	4,290	--	1.49
Central Iowa 2	650	1 SSB - 1 VIB - 1 VTA	1.07	560	0.32	0.22
Northwest Iowa 1	1,400	1 SSB - 1 VTA	2.91	3,710	--	1.68
Northwest Iowa 2	4,000	1 SSB - 1 VIB - 1 VTA	2.96	1,120	1.01	0.60
Southwest Iowa 1	2,300	1 SSB - 10 VTA	7.49	11,550	--	4.05
Southwest Iowa 2	1,200	1 SSB - 1 VTA	3.72	6,275	--	3.44

Central Iowa 1 (CN IA 1) was a 3.09 ha feedlot permitted for 1,000 head of cattle. Runoff effluent drained into a solid settling basin designed to hold 4,290 m^3 of effluent. The VTA consisted of two channels operated in parallel; each channel was 24 m wide and averaged 311 m long. Central IA 1 VTA soil consisted of Clarion loam, Cylinder loam, and Wadena loam (Soil Survey Staff, NRCS USDA, 2010). The VTS at Central Iowa 2 consisted of a SSB, VIB, and VTA. Runoff from the 1.07 ha feedlot drained into a concrete SSB which released effluent into a 0.32 ha VIB. Effluent captured in VIB tiles was pumped onto a VTA. Soils in the VIB consisted of Nicollet loam and Webster clay loam and the VTA was Harps loam (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 1 (NW IA 1) consisted of a 2.91 ha feedlot permitted to hold 1,400 head of cattle. Feedlot runoff was collected in a SSB with a volume of 3,700 m^3. The SSB outlet pipe discharged onto VTA consisting of Galva silty clay and Radford silt loam soils (Soil Survey Staff, NRCS USDA, 2010). Northwest Iowa 2 (NW IA 2) had an SSB-VIB-VTA system designed to control runoff from a 2.96 ha concrete feedlot. A settling basin collected the feedlot runoff and released it to a 1.01 ha VIB drained by 15 cm diameter perforated tiles installed 1.2 m deep and spaced 4.6 m apart. Flow from the tile lines was collected in a sump and pumped onto the VTA divided into two 27 m wide channels. The channel receiving effluent was switched manually by the producer. Northwest IA 2 consisted of Moody silty clay loam (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 1 (SW IA 1) was a 7.49 ha feedlot with an 11,550 m^3 solid settling basin that released effluent to a 4.05 ha VTA was divided into ten channels. Tile lines, installed to control water table depth below the system and enhance infiltration of effluent into the soil, surrounded each of the VTA channels. Soils in the VTA consisted of mostly Judson silty clay loam and smaller areas of Colo-Ely complex (Soil Survey Staff, NRCS USDA, 2010). Southwest Iowa 2 (SW IA 2) was a 3.72 ha feedlot. Runoff drained into a solid settling basin and was released to a 3.44 ha VTA constructed with earthen berm level spreaders along the length. The spreaders slowed the flow of effluent through the system, increasing the time for infiltration and promoting sedimentation of particulates suspended in the flow. Southwest IA 2 VTA soil consisted of Kennebec silt loam (Soil Survey Staff, NRCS USDA, 2010). At each site grass areas of the same soil series were found and sampled to evaluate the soils labile carbon and nitrogen content of soil not receiving the effluent application; these properties are thought to represent the original site conditions prior to use of the vegetative treatment system, and thus provide an opportunity to evaluate the impact of five years of runoff effluent application on soil labile organic matter.

Soil Sampling

At each of the six sites five soil samples were collected from the vegetative treatment area and five more from a paired area that did not receive the feedlot runoff effluent application. This sampling methodology was utilized as soil sample collected before vegetative treatment construction and use were not available. Each soil sample was collected by compositing soil from five randomly selected locations within the vegetative treatment area or paired area; at each sampling location a push-probe was used to collect soil to a depth of 15.2 cm (6 inches) from twenty spots within a 1.5-m radius of the selected location. This sampling methodology was used to minimize the within treatment component variability due to differences in greater phosphorus loading near settling basin inlets and variability in soil properties over the relatively large vegetative treatment areas. Collected soil was placed in a plastic bag, placed on ice, and brought back to the Agricultural Waste Management Lab at Iowa State University. Once back the soil samples mass was determined and they were spread out on trays to air dry. Aggregates were crushed and sieved to pass a screen with 2 mm openings. Rocks and visible vegetation were removed during the sieving process. The mass of soil passing and retained on the 2 mm screen was determined to estimate the amount of course fraction present in each soil and determined the moisture content of the soil. A subsample of the soil passing the 2 mm screen was dried in an oven at 105°C for 24 hours to determine the air dried moisture content of the soil. The remaining soil was placed in screw-cap plastic bottles. A subsample of this soil was used in the biological fractionation procedure.

Biological Fractionation Procedure

Biologically available carbon and nitrogen were determined using long-term (>1 year) laboratory incubations with repeated leaching (Stanford and Smith, 1972). Although other fractionation techniques, physical and chemical can be used to separate organic matter pools, we were interested in determining whether nitrogen and carbon being added to the vegetative treatment area was being stored in biologically available pools or if it was stored in forms that were not available. Laboratory incubations provide the only fractionation method that directly assays this question (Robertson and Paul, 1999) and as such were utilized in this study.

A 100 gram (air-dried weight) subsample of each soil sample was incubated at optimal temperature (35°C) (Campbell et al. 1993; Drinkwater et al., 1996) for approximately 1-year to determine the biologically available carbon and nitrogen. A plastic (Buchner funnel was used to contain the soil during the incubation. A glass fiber filter (Whatman GF/A, Whatman Inc., Ann Arbor, MI) and an "extra thick" glass fiber prefilter were placed at the bottom of the funnel. Glass wool was placed on top of the filter and then the 100 grams of soil was added. This glass wool was added to help provide structure and keep the soil at aerobic conditions. A third glass fiber filter (Whatman GF/A, Whatman, Inc., Ann Arbor, MI, USA) was placed on top of the soil to avoid particle dispersion during water and leaching solutions additions (Motavalli et al., 1995). The Buchner funnel units were then placed into individual air-tight plastic containers (0.83 L volume) with screw top lids. Each lid had been fitted with a septa to allow gas samples to be drawn from the head space. Water holding capacity (determined by saturating the soil and then allowing it to free day two hours) was determined from on a separate subsample of soil. Distilled water was added to the soil being incubated to bring it to 70% of its field capacity. Soil mass was then tracked and additional water added as necessary (every few days) to make sure the soil sample remained at 70% of its field capacity as this moisture content maximizes nitrification (citation).

The labile carbon pool size was estimated by capturing carbon dioxide in the headspace of the incubation jars. Soil respiration rates were measured on days 1, 3, 6, 10, 15, 19, 27, 33, 39, 46, 56, 60, 67, 74, 88, 96, 104, 110, 119, 136, 147, 160, 172, 186, 199, 216, 238, 261, 277, 302, 327, 357, and 385 (33 times). On dates when respiration was measured samples were fanned with ambient air for 15 minutes. The covers were then screwed onto the samples to make them air tight and carbon dioxide allowed to accumulate. Samples were kept sealed for periods ranging from several hours (beginning of the incubation) to several weeks (end of the incubation). A series of eight blanks, kept with the soil samples at all times, where also sealed at this time. After the rescribed amount of time had passed carbon dioxide levels present in the head space of the blanks and soils samples was measured using an infrared gas analyzer (LICOR-3200; LICOR, Lincoln, NE, USA). During headspace gas sampling a 10 mL air-tight syringe was used to stir gasses in the head space by drawing and reinjecting five samples. A sample of the head space gas was then drawn and injected into the gas analyzer which measured the mass of CO_2 present in the sample. The mass of CO_2 present and the volume of the sample was recorded, allowing calculation of the concentration of CO_2 in the headspace gas. The average concentration of CO_2 in the eight blanks were used to correct CO_2 production to account for ambient levels. The mass of CO_2 respired by the soil was then calculated multiplying the change in headspace concentration by the volume of the container (corrected for the volume occupied by the Buchner funnel, the soil material, and the water within the container). The mass of CO_2 generated was then divided by the mass of dry soil and the length of time the sample was covered to normalize the results to mass of CO_2 per mass of soil per unit time. This respiration rate represented the rate at the midpoint of the time interval the sample was covered. Respiration rate data was then fit to a two pool decaying exponential model (shown as equation) using a least squares fitting procedure. In this equation R_r is the soil respiration rate (mg C/kg soil-day), C_l and C_r are the size of the easily mineralizable and slowly mineralizable pools respectively (mg C/kg soil), k_l and k_r of the rate constants associated with these pools (day-1), and t is the incubation day (day). The equation was then integrated with respect to time to estimate the total mass of carbon respired over the course of the incubation, which was assumed to represent the biologically available fraction.

$$R_r = C_l k_l \exp(-k_l t) + C_r k_r \exp(-k_r t) \tag{1}$$

Labile nitrogen pool sizes were estimated by extracting mineral nitrogen from the soil sample with periodic leaching. Leaching occurred on days 0, 7, 21, 35, 50, 75, 101, 151, 250, and 385. The soil was leached with a solution containing all essential nutrients except N (Stanford and Smith, 1972; Nadelhoffer, 1990). At each leaching ~100 mL (exact volume was measured by measuring density of the leaching solution and by weighing a graduated cylinder with the leaching solution in it and after the leaching solution was added to the soil sample) of N-free solution was added to the top of the filter, allowed to equilibrate with the soil for 0.5 hours, and then drawn thro ugh the soil with a weak vacuum. The vacuum was applied until leachate stopped dripping from the filter (usually around 5 minutes, but always less than 10). Leachate was frozen until conclusion of the year long incubation. The frozen leachate was then thawed and analyzed for NH_4^+-N and NO_3^--N concentrations by steam distillation and trapping in a boric acid solution and titration (for NH_4^+-N) and an ion specific electrode (for NO_3^--N). The volume of leachate was determined by measuring the mass of leaching solution added to the soil sample and the mass of the soil sample before and after leaching, and measuring the density of the leachate. The mass of NH_4^+-N and NO_3^--N leached was calculated by multiplying the measured concentrations by the volume leached. This value was normalized by dividing the mass of NH_4^+-N and NO_3^--N leached by the air-dry weight of the soil sample. This data was then fit to a single pool exponential model of the cumulative mass leached. The labile N pool was defined as the sum of all inorganic N in the leachate solutions. A single pool model (2) was fit to the cumulative nitrogen mineralization data using a least squares fitting procedure. In this equation R_N is the soil nitrogen mineralization rate (mg N/kg soil-day), N is the size of the mineralizable pool (mg N/kg soil), k is the rate constants associated with this pool (day-1), and t is the incubation day (day). Differentiation of this equation provides the rate of nitrogen mineralization as a function of time.

$$R_N = N(1 - \exp(-kt)) \tag{2}$$

Analysis

The mass of labile carbon, nitrogen, and the C:N ratio of the labile organic matter mineralized from the VTA soils and the paired grass area were compared using t-tests on the total mass of carbon and nitrogen mineralized. Visual comparison of the carbon and nitrogen mineralization curves were made to evaluate if differences in the relative recalcitrance of the organic matter existed. That is, the mineralization curves were inspected to evaluate if samples from the different treatment (VTA versus grassed area) had the same shape or if the curves differed for different portions of the incubation. This analysis was supplemented with t-tests conducted at every measured respiration point and every cumulative mineralization point to evaluate if observed differences were significant.

RESULTS AND DISCUSSION

Rates of carbon respired from the soil pool declined rapidly during the first 15 days of the incubation and then exhibited a slower linear decline for the next 150-200 days before becoming relatively stable at around day 180-220 (figure 1). At most of the sites (CN IA 1, NW IA2, SW IA 1, and SW IA 2) the trends in carbon mineralization from the grassed area soil and from the vegetative treatment area soil were essentially the same shape although at all four of these sites respiration from VTA soils tended to be slightly higher on average than soils from the grass area. These result was seen to a much greater extend at NW IA 1 where vegetative treatment area soil exhibited a much greater respiration rate than the paired grass area soil for the first 165 days of the incubation. At this point the two curves became very similar; however, the vegetative treatment area soil still tended to have slightly higher respiration rates. At CN IA 2 the opposite trend was found. At this site the paired grass area soil tended to have higher respiration rates throughout the incubation with the first 200 days being the period where this was most evident. This site utilized a vegetative infiltration basin in the treatment system prior to applying effluent to the vegetative treatment area. This component was extremely effective at removing many contaminants including oxygen demanding substances and solids and thus presumable greatly reduced the organic loading onto the VTA at this site. The other sites did not utilize vegetative infiltration basins or, as was the case at NW IA 2, the infiltration wasn't nearly as successful at removing contaminants prior to effluent application onto the vegetative treatment area due to the use of a surface drain to expatiate drainage of the vegetative infiltration basin.

At all sites the carbon respiration data were well fit by the double exponential decay model with R^2 values ranging from 0.983to 0.995. This indicates that utilizing the fitted equation to estimate respiration rates on dates when it was not measured and using it to determine the mass of carbon respired (by integrating to determine the area under the curve) should be appropriate. Although caution must be exercised in evaluating the meaning of the fitting parameter determined (Table 2), the fact that total mass of respired carbon represents matches (95-99%) the sum of the easily mineralizable and slowly mineralizable pools ($C_l + C_r$) indicates that the fitted parameters are reasonable and can be interpreted.

We had originally hypothesized that using the soil as vegetative treatment areas would increase mineralizable carbon. This hypothesis follows from Stewart et al.'s (2007) description of carbon saturation where a hierarchy of carbon storage (in pools of differing recalcitrance due protection mechanisms –physical protection, chemical protection, biological recalcitrance, and non-protected) is proposed. In this model they suggest increasing inputs of carbon can cause increases in biologically available carbon will increase, although only slowly if the soil is near its carbon saturation. Thus we expected increases in soil carbon due to the large increase in carbon loading from application of feedlot runoff, but recognized that these change could be small as Iowa soils e typically carbon rich and presumably near their carbon saturation limit.

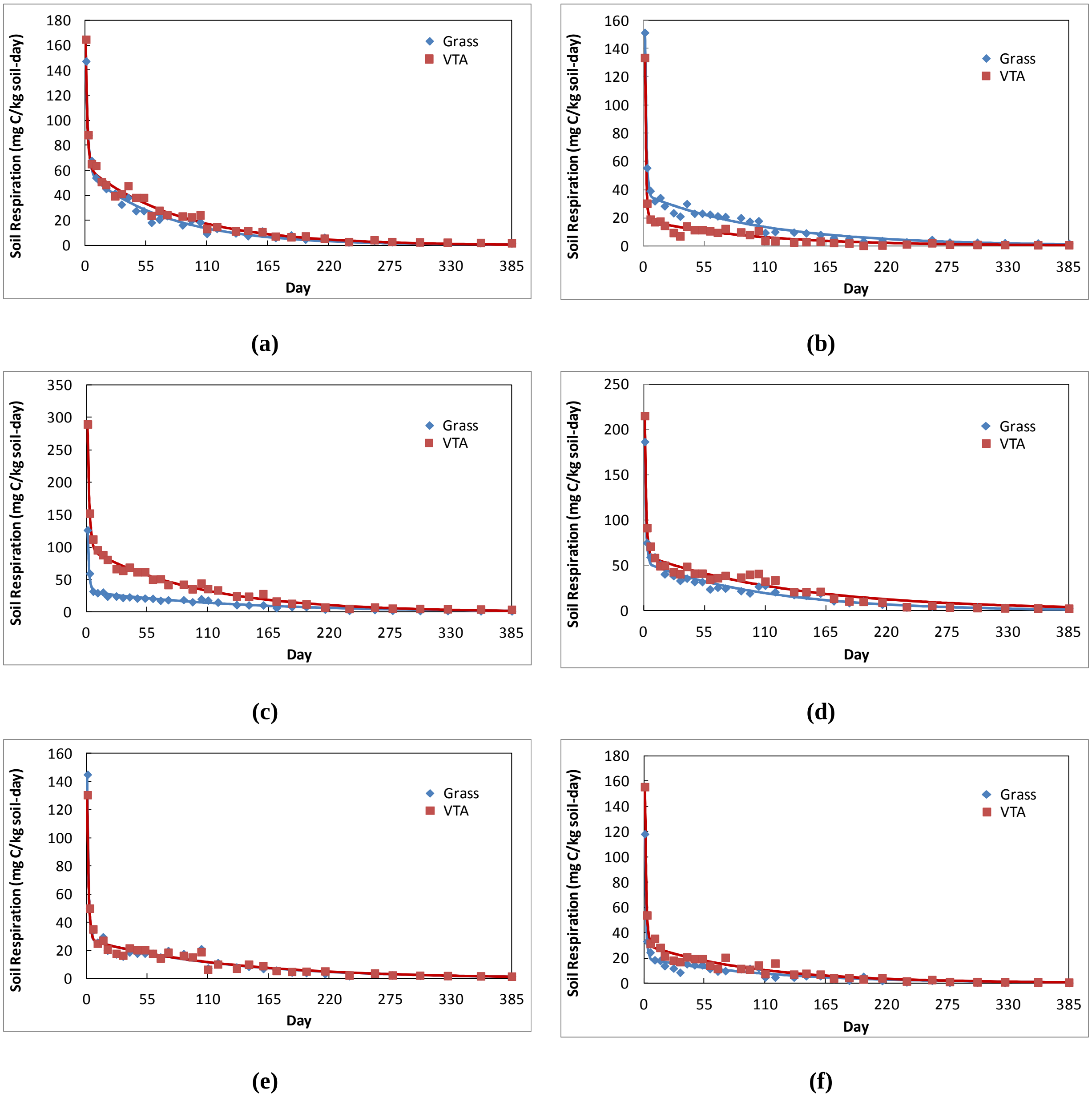


Figure 1. Soil respiration rates (mg C/kg soil-day) concentrations as a function incubation date for the grassed and VTA soil at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident. Solids lines represent model fit of the data.

As can be seen, the size of the easily mineralizable pools at CN IA 1 and CN IA 2 did not change, although in both cases the lability of these pools did increase slightly. NW IA 1 experienced a sizeable increase in this carbon pool (318 mg C/kg soil), but the lability was slightly decreased. Similar changes (i.e., increased pool size with decreased lability) were noted at NW IA 2 and SW 1, although in these cases increases in carbon pool sizes were much smaller. At SW IA 2 a small decrease in carbon pool size was found. All sites except CN IA 2 had increased slowly mineralizable carbon pool sizes. This pool was much larger than the quickly mineralizable (6 – 19x) and rate constants where much more similar among the sites, with only small changes in rate constants seen. At both northwest Iowa sites these pools showed large responses, approximately doubling in size after five years of use as vegetative treatment areas. Responses at CN IA 1and and 1 were small with increases of approximately 20% occurring. Little to no change was seen at SW IA 2 while CN IA 2 decreased by approximately 50%. Although it is unclear what this pool represents, it appears that this pool may be able to accumulate more carbon depending on its current pool size in relation to its saturated capacity; however, more research is needed to understand this pool. Thus, in general these results were similar to what we expected as most sites exhibited an increase in mineralizable carbon; the one exception to this was at CN IA 2 where mineralizable carbon levels declined. We hypothesize that this was due to the application of large amounts of the dilute (due to treatment in the highly effective vegetative infiltration basin) wastewater.

Table 2. Summary of the rate coefficients determined for the two-pool carbon respiration model.

Site	Treatment	C_l (mg C/kg soil)	C_r (mg C/kg soil)	k_l (day^{-1})	k_r (day^{-1})
CN IA 1	Grass	302	4374	0.467	0.013
	VTA	299	5404	0.651	0.012
CN IA 2	Grass	316	3979	0.863	0.009
	VTA	315	1804	1.103	0.010
NW IA 1	Grass	270	4413	0.729	0.007
	VTA	588	9924	0.579	0.010
NW IA 2	Grass	365	5917	0.858	0.009
	VTA	435	8305	0.753	0.007
SW IA 1	Grass	268	2241	0.952	0.009
	VTA	346	3084	0.816	0.010
SW IA 2	Grass	326	3469	0.788	0.008
	VTA	292	3553	0.720	0.008

Nitrogen mineralization rates remained approximately linear for the first 100-150 days of the soil incubation. At that time rates began to slow and a plateau can be seen in the cumulative nitrogen mineralization curves shown in figure 2. Interestingly, nitrogen mineralization was greater in all the vegetative treatment area soils than in the paired grass area, including CN IA 2. This result was not unexpected as these waste treatment systems receive high nitrogen application rates (593-1866 kg

N/ha-yr) as they are being utilized as effluent disposal areas. The mineralization pattern at Central Iowa 2 is particularly interesting, at this site we saw reduced carbon mineralization and thus had expected lower nitrogen mineralization as carbon and nitrogen retention and storage mechanisms are often linked. However, looking closely at the nitrogen mineralization patterns at this site it appears that what may have occurs is that the reduced mineralizable carbon within the soil reduced the soils ability to recycle nitrogen in microbial biomass during the incubation and thus allowed nitrogen leaching earlier in the incubation than from the paired grass area. In this incubation nitrate accounted for 96.1 ± 1.4% (ave ± s.d) of the total mass of nitrogen leached in these incubations. The first two leaching periods, day 0 and day 7, averaged only 87 and 91% of the nitrogen leached as nitrate, but the 8 other leaching (day 21, 35, 50, 75, 101, 151, 250, and 385) averaged 95-97% of nitrogen in the nitrate form.

At all sites the nitrogen mineralization data was well fit by the one-pool exponential decay model with R^2 values ranging from 0.990 to 0.999. This indicates that utilizing the fitted equation to estimate cumulative nitrogen mineralization on dates when it wasn’t measured and to determine the rate of nitrogen mineralization (by differentiation to determine the slope of the curve) should be appropriate. Although caution must be exercised in evaluating the meaning of the fitting parameter determined (table 3), the fact that cumulative mineralized nitrogen represents 88-99% of the mass estimated in the mineralizable pool (N) again indications that the fitted parameters are reasonable. At all sites, except CN IA 2, the size of the mineralizable nitrogen pool was substantially larger in the vegetative treatment area soil than its paired grass counterpart. In particular CN IA 1, NW IA 1, and NW IA 2 all had statistical significantly larger nitrogen pools ($p < 0.001$, <0.001, and 0.018 respectively). While not quite significantly different SW IA 1 and SW IA 2 also tended to show highly increased nitrogen levels ($p = 0.078$ and $p = 0.130$). From a practical view the mineralizable nitrogen pool size more than doubled at NW IA 1, increased by about 50% at CN IA 1 and NW IA 2, and increased by 20% at SW IA 1 and 2. At CN IA 2 the pool size remained constant despite the fact that carbon pool sizes decreased. No consistent trend in nitrogen lability was seen as most sites remained relatively similar; however, it appears that lability of nitrogen at CN IA 2 may have increased. We believe this may have occurred due to the loss of carbon; since less mineralizable carbon was available present to maintain microbial biomass the nitrogen cycle was looser and nitrogen could be leached earlier in the incubation.

Table 3. Summary of the mineralization rate constant (k) and the mineralizable nitrogen pool size (N) determined for the one-pool nitrogen mineralization model.

Site	Treatment	N (mg N/kg soil)	k (day^{-1})
CN IA 1	Grass	504	0.0075
	VTA	753	0.0085
CN IA 2	Grass	385	0.0074
	VTA	369	0.0111
NW IA 1	Grass	464	0.0077
	VTA	1041	0.0059
NW IA 2	Grass	611	0.0054
	VTA	906	0.0069
SW IA 1	Grass	385	0.0086
	VTA	457	0.0101
SW IA 2	Grass	465	0.0075
	VTA	536	0.0080

Finally, we used these data to evaluate the C:N ratio of the labile pool. This was done in two ways; first by plotting the cumulative mass of nitrogen mineralized against the cumulative mass of carbon respired (figure 3) and second by evaluating how the C:N ratio of the labile pool changed through the incubation. This analysis provides insight into the C:N ratio of the mineralized organic matter on average, but not how it varies throughout the incubation. In this analysis we found that the C:N ratio of the organic matter on average was 13.7. Moreover, the two pools showed a high degree of correlation as the size of the labile carbon pool explain over 80% of the variation in the size of the labile pool. However, a closer look indicates that the labile nitrogen pools have been enriched in comparison to the paired grassland soil counterparts, that is they consistently plotted above the best-fit line in Figure 3 while soils from the grassed area typically plotted below the line. This is an interesting phenomenon and is one of the traits often associated with nitrogen saturation in forest ecosystems. As this is the case it would seem to indicate that the vegetative treatment areas are progressing to greater degrees of nitrogen saturation which may make them more susceptible to future nitrogen loss.

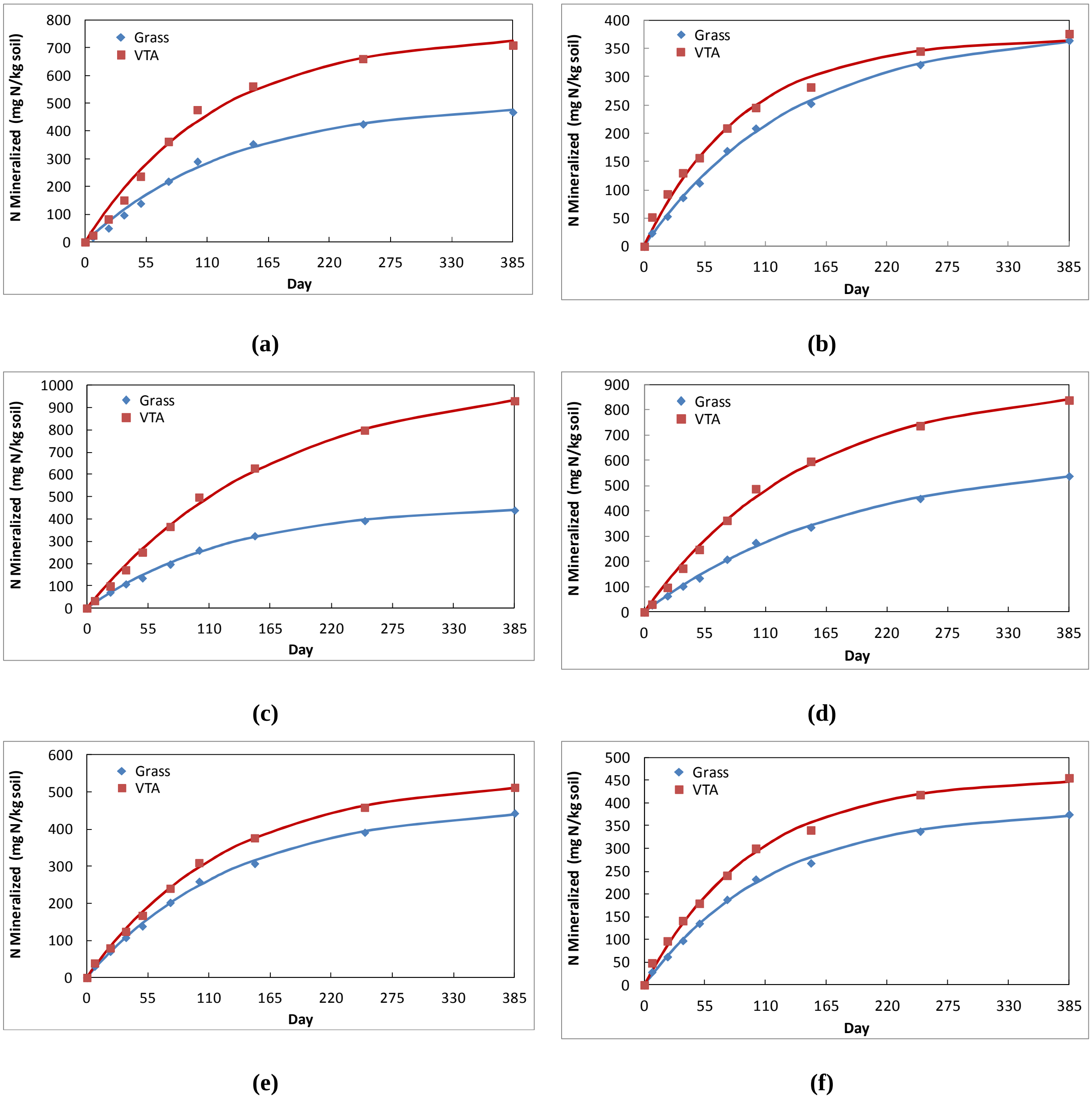


Figure 2. Cumulative nitrogen mineralization masses leached (mg N/kg soil) as a function incubation date for the grassed and VTA soil at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident. Solids lines represent model fit of the data.

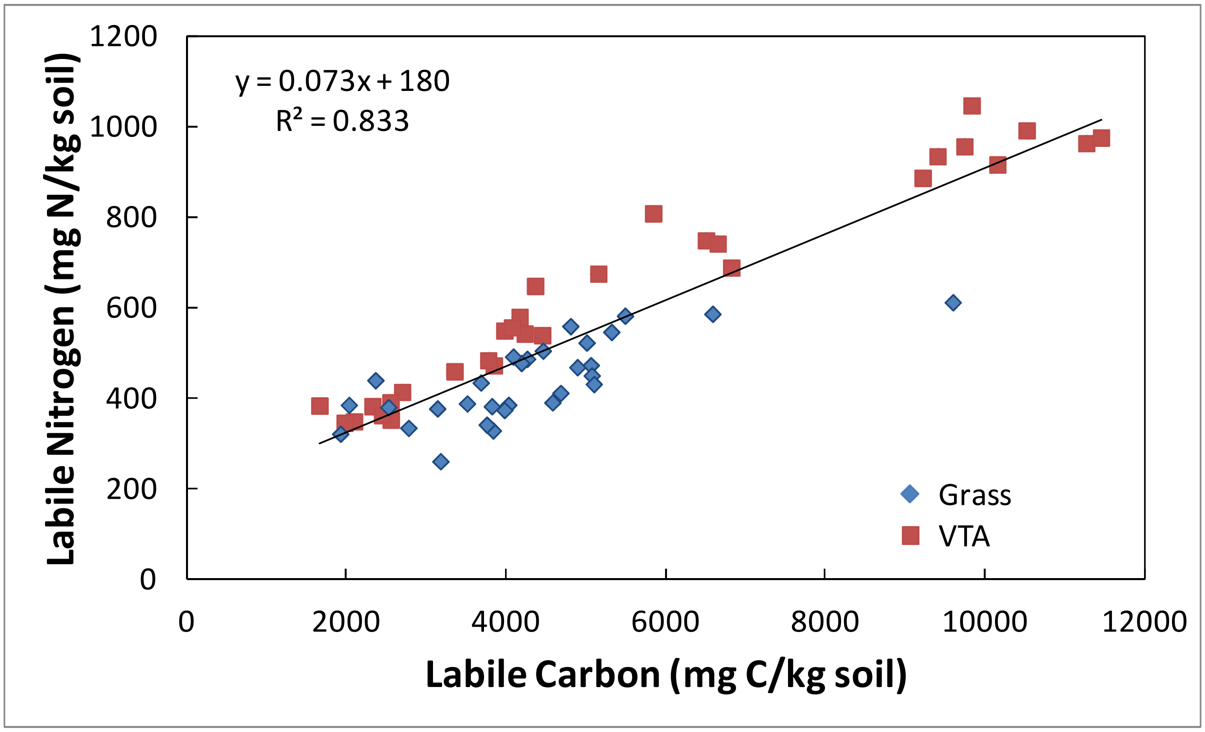
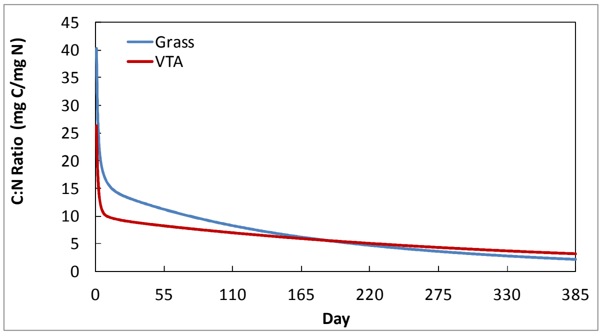


Figure 3. Labile nitrogen versus labile carbon of the 60 soil samples tested. Results showed that for every unit increase in labile nitrogen an increase of 13.7 units of carbon was expected; however, it also indicated that labile organic matter in the vegetative treatment area soils is nitrogen enriched.

The carbon respiration rate and nitrogen mineralization rate were calculated for every day of the incubation using the fitted model equations. The ratio of carbon mineralization to nitrogen mineralization was then plotted as a function of time. This analysis was performed to evaluate how the characteristics of the labile organic matter changed through the incubation and to determine if differences in characteristics were seen between the vegetative treatment area and grassed area soils. In general, the largest C:N ratios were seen near the beginning of the incubation. These values quickly (usually less than 50 days) decreased to ratios of around 10 or in some cases slightly lower. At both CN IA 1 and CN IA 2 the labile organic matter fractions during the early stages of the incubation (before 165 and 330 respectively) appeared to be nitrogen enriched in the VTA soil. As time progressed the labile organic matter C:N ratios became more similar between the two soils. Similar trends were noted at NW IA 2 while SW IA 1 and SW IA 2 VTA and grassland soils showed basically the same trends throughout the incubation. NW IA 1's soil showed the opposite trend; during the initial period of the soil incubation the soil appeared to have more easily respired carbon than the mass of nitrogen mineralized would have suggested. As the incubation progresses though the C:N ratios narrow rapidly to ratios values as low as 4 for the VTA soil.



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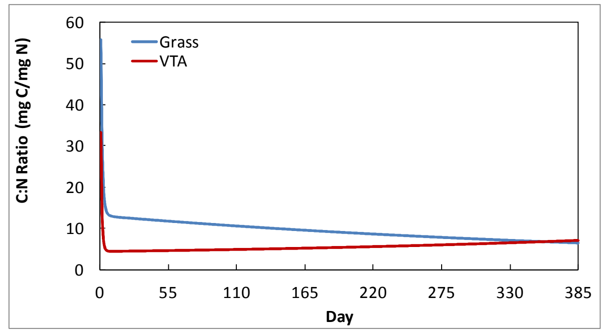

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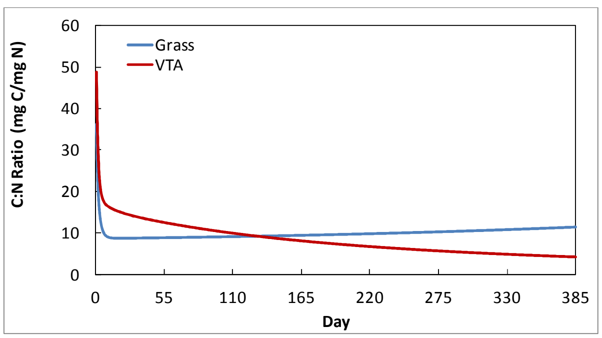

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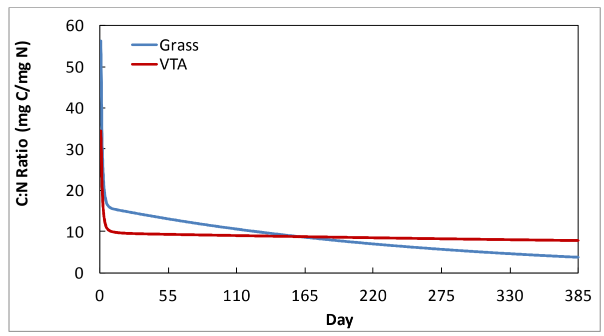

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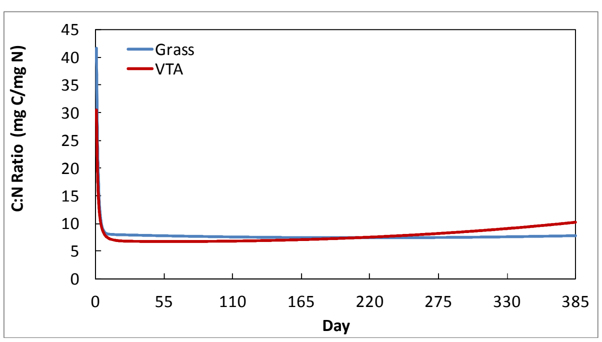

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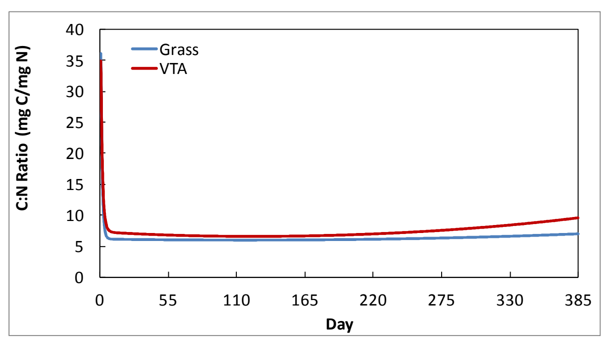

(f)

Figure 4. Carbon-to-Nitrogen ratios as a function incubation date for the grassed and VTA soil at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

Statistical analysis of the mass of C respired, N mineralized, and the C:N ratio of the labile organic matter is presented in Table 4. At Central Iowa 1 the increase in labile carbon in VTA was not quite significant ($p = 0.060$), but the increase in mineralizable N was ($p < 0.001$) and a significant enrichment of nitrogen was present in the labile organic matter ($p = 0.017$). At Central Iowa 2 a significant decrease in mineralizable carbon was monitored ($p < 0.001$) while no change in mineralizable nitrogen was seen. This indicates that a drastic and significant enrichment of nitrogen occurred in the labile pool. At Northwest Iowa 1 significant increases in mineralizable carbon and nitrogen occurred; however this results in an increase in C:N ratio of the labile organic matter. This result is particularly interesting as the data points from this site show a strong indication of nitrogen enrichment in figure3, i.e. VTA points are above the regression line while soil samples from the paired grass are below the regression line. At Northwest Iowa 2 a significant ($p = 0.018$) increase in mineralizable nitrogen was seen, a non-significant increase in mineralizable carbon was also observed ($p = 0.200$) as was a non-significant increase in nitrogen enrichment of the labile organic matter ($p = 0.154$). At Southwest Iowa 1, increases in mineralizable carbon, nitrogen, and a greater carbon to nitrogen ratio were observed, but none of the changes were significant. At Southwest Iowa 2 none of the changes in mineralizable carbon, mineralizable nitrogen, or carbon-to-nitrogen ratio were significant ($p = 0.902$, 0.130, 0.058 respectively).

CONCLUSIONS

A biological fractionation technique was used to evaluate if five years of use had caused a significant increase in biologically available soil carbon and nitrogen, or nitrogen enrichment of labile soil organic matter. The results indicated that increases in biologically soil carbon did occur, but increases in nitrogen were typically larger and often more significant. This has resulted in nitrogen enrichment of the labile soil organic matter, a typical symptom of nitrogen saturation. This has resulted in lower carbon-to-nitrogen ratios in the soil organic matter, reducing its ability to serve as a nitrogen sink. This could indicate that the vegetative treatment areas are becoming more prone to nitrogen loss either through leaching or through gaseous emission as the capacity of the soil to utilize and retain nitrogen is becoming exhausted.

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Table 4. Summary of mass of C respired, N mineralized, and the C:N ratio of labile organic matter at Central Iowa 1 (CN IA 1), Central Iowa 2 (CN IA 2), Northwest Iowa 1 (NW IA 1), Northwest Iowa 2 (NW IA 2), Southwest Iowa 1 (SW IA 1), and Southwest Iowa 2 (SW IA 2).

Site		C Mineralized mg C/kg soil	N Mineralized mg C/kg soil	C:N Ratio mg C/mg N
CN IA 1	Grass	4718	475	10.0
		(309)	(60)	(1.3)
	VTA	5701	724	7.8
		(955)	(64)	(1.0)
	p-value	0.060	<0.001	0.017
CN IA 2	Grass	4186	362	11.6
		(858)	(78)	(0.5)
	VTA	2099	364	5.8
		(307)	(18)	(0.9)
	p-value	<0.001	0.954	<0.001
NW IA 1	Grass	4299	440	9.9
		(677)	(90)	(0.8)
	VTA	10296	935	11.0
		(1032)	(36)	(0.8)
	p-value	<0.001	<0.001	0.045
NW IA 2	Grass	6235	536	11.5
		(2011)	(78)	(2.5)
	VTA	8270	844	9.7
		(2566)	(219)	(0.9)
	p-value	0.200	0.018	0.154
SW IA 1	Grass	2444	371	6.6
		(510)	(47)	(1.5)
	VTA	3335	448	7.4
		(721)	(71)	(0.6)
	p-value	0.054	0.078	0.339
SW IA 2	Grass	3651	440	8.3
		(731)	(57)	(1.0)
	VTA	3707	512	7.2
		(663)	(76)	(0.5)
	p-value	0.902	0.130	0.058

Chapter 10. Vegetative Treatment Systems: Design, Management, and Siting Recommendations

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Abstract. *Runoff from open lot animal feeding operations has been recognized as a potential pollutant to receiving surface waters. This effluent is known to contain nutrients such as nitrogen and phosphorus, as well as other potential pollutants such as organic matter, solids, and pathogens. Due to increased recognition of the potential impacts feedlots can have on water quality, cattle producers are facing increasing pressures to improve their feedlot runoff control systems. As a result, vegetative treatment systems are increasingly being utilized on beef feeding facilities; thus producers are seeking guidance on what it takes to make these systems successful. This has lead to considerable research over the last ten years on how to successfully utilize vegetative treatment system. Here, we provide recommendations, based on our experiences utilizing them on concentrated animal feeding operations, about what it takes to make these systems successful. In general, we have found that it requires a combination of proper siting, sound design, and good management to get ideal performance. It is critical that minimum distances (~1.5 – 3 m) to groundwater are maintained to ensure system releases do not occur except from events larger than the design storm. Vegetative treatment system designs should seek to provide flexibility so that the operator can adapt to current weather conditions. To achieve this we recommend building large settling basins with controlled outlets that allow the producer to hold effluent for several days until weather conditions permit application to the vegetative components. Finally, attentive management is required. Producers must be vigilant in operating the system to ensure channeling doesn't develop, that effluent is being applied at a rate the system is capable of handling, and making sure that systems components are operating effectively. Finally, the current state of the knowledge on feedlot runoff control is accessed and recommendations about areas requiring further research provided. We feel future research on evaluating nitrogen emissions and maximizing denitrification are imperative for evaluating these systems.*

Keywords. *Runoff, vegetative treatment systems, runoff control, open lot, vegetative treatment area*

Introduction

Runoff from open-lot animal feeding operations (AFOs) has long been recognized as a potential pollutant to receiving waters. As such, adequate control and management of this wastewater is required to mitigate its potential impacts on surface and ground waters. Traditionally, the level of control implemented at animal feeding operations has varied by the size and geographic location of the operation. Typically, larger farms (concentrated animal feeding operations, i.e., those over 1000 head of cattle) have used containment basin systems to control runoff while small farms (< 300 heads) have often relied on solids settling systems and vegetative filtering to manage their environmental risk. However, changes in United State Environmental Protection Agency's (US EPA) effluent limitation guidelines (ELGs) that allowed concentrated animal feeding operations (CAFOs) to utilize alternative systems, i.e., systems other than containment-land application systems, if the performance obtained was as good as or better than that of a traditional system have lead to greater interest in the use of alternative runoff control technologies. Similarly, increasing scrutiny over the impact small and medium sized (< 1000 head) operations can have on water quality has spurred the development and implementation of cost effective options of managing lot runoff.

Vegetative treatment systems are one option that has shown good potential to provide runoff control and are less costly to construct than containment basins (Bond et al., 2011).As a result, many operations are seeking guidance on how to site, design, and manage vegetative treatment systems to be successful. Thus far, guidance on these topics has typically been based on either the review of vegetative treatment systems performance by Koelsch et al. (2006) or the companion resources developed for the Heartland Regional Water Coordination Initiative Vegetative Treatment System Guidance document (USDA NRCS, 2006). Although these references remain useful, the last ten years has seen installation and intensive monitoring of a significant number of vegetative treatment systems at commercial feedlot operations. This has resulted in significant new knowledge about how treatment system performance varies under different conditions, what it takes to make these systems successful, and raised new questions about what is required to ensure these systems are appropriate and remain effective. Thus, it seems appropriate that vegetative treatment systems again be reviewed to ascertain the current state-of-the-knowledge of vegetative treatment systems, specifically focusing on what the last ten years of field implementation has taught us about maximizing their performance and sustainability.

The format of this review is as follows: (1) a discussion of current government regulation regarding control of feedlot runoff, (2) a review of vegetative treatment system basics (terms and definitions, system configurations, component types, system variations, etc.), (3) a review of the design principles and goals of vegetative treatment systems (theory behind design, the purpose of different treatment components, and hydraulic, nitrogen, and phosphorus cycling and retention in vegetative treatment systems siting, design, and management). Throughout each section we will try to highlight the current state of the knowledge of the topic, provide tips related to design, siting, and managing the runoff control systems, and address areas that could be the topic of future research.

Feedlot Runoff Regulations

Runoff from open-lot animal feeding operations is regulated by a combination of federal and state guidelines. Federal regulation can be traced to the Federal Water Pollution Control Act Amendments of 1972, which placed the U.S. Environmental Protection Agency (EPA) in charge of developing runoff control guidelines. As a result, the EPA developed the Effluent Limitation Guidelines (ELGs), which set criteria for when an animal feeding operation could be designated as concentrated animal feeding operations, i.e., CAFOs, and specified the design and operating standards required of waste management systems at these facilities (Anschutz et al., 1979). Historically, these ELGs required collection, storage, and periodic land application of manures and wastewaters; however, modifications to the ELGs in 2003 (Federal Register, 2003) and reaffirmed in 2008 (Federal Register, 2008) allowed the use of alternative treatment technologies if performance was equivalent to, or exceeded, that of the traditional containment-land application systems (EPA, 2006). Runoff control requirements for open lot animal feeding operations not designated as CAFOs are typically set by state regulations. In Iowa, non-CAFO open lot cattle feeding operations are subject to the regulatory requirements of Chapter 65 of the Iowa Administrative Code (Iowa DNR, 2007), which requires removal of settlable solids from the runoff and that any settled effluent released from the runoff control system not cause the receiving water body to exceed the water quality criteria set for its designated use. Similar standards, ranging from the Iowa standard, i.e., no harm to waters of the state, to a “no potential to discharge” requirement exist in other Midwestern states; for instance Nebraska requires no hydrologic connection between the feedlot and surface water while Minnesota requirements range from buffer strips to no potential to discharge depending on the environmental risk runoff from the operation poses (Branch, 2003).

Although not explicitly stated in the EPA's ELGs, the preamble to the 2008 guidelines insinuates that the purpose of these standards should be to ensure surface waters are meeting applicable water quality standards (EPA, 2008). Specifically, the 2008 ELG preamble suggests that where technology-based limitations prove insufficient to attain or maintain applicable water quality standards; NPDES permits must contain more stringent limitations representing that level of control necessary to ensure that receiving waters meet applicable water quality standards. In attempting to ensure surface waters meet water quality standards the EPA has predominately focused on point sources. This has resulted in great progress, but in many agricultural watersheds water quality standards are still not being met. Researchers, government officials, and the general public have come to the realization that we must now focus our efforts on nonpoint source pollution to achieve the desired concentration standards. Thus far legislation has focused on the "point source," i.e., the production area, but to better evaluate the overall impact animal feeding operations have on water quality, nutrients releases from both the production area (animal housing, feed and manure storage areas) and the land application are need to be considered.

This intention of the EPA to move towards this whole-farm evaluation of AFOs is made clear in the preamble of the CAFO bill. The preamble states "regulatory provisions are targeted toward the CAFO's wastewater discharges, but EPA encourages operations electing to participate in the alternative performance standards program to consider environmental releases holistically, including opportunities for achieving improvement in multiple environmental media (Page 7222, emphasis added). Unfortunately, given the structure of the CAFO rules provisions, EPA requires that the baseline level of wastewater discharges to be considered when evaluating the performance of an alternative system be limited to those discharges from the production area only stating in the final rule that the alternative manure treatment system should have on net, no additional discharge as compared to traditional containment systems (EPA, 2003). This partial treatment of a CAFOs manure management system, splitting off the water quality performance of its production area subsystem from that of its land application subsystem is also inconsistent with EPA's own characterization of land application areas as being "integral to CAFO operations" (EPA, 2006). Still, the practical implication of the rulemaking language, despite the encouragement given in the preamble and EPA's related statements, precludes a multi-media approach when it comes to the how differing production area wastewater management systems might contribute to performance improvements in the land application system. Given these statements in the CAFO rule preamble, a significant opportunity exists to provide a framework to make the comparison between baseline waste management systems and proposed alternative treatment systems. This framework must consider how nutrient management at the farm and watershed scale would be impacted by the change in waste management practice, i.e., if an advanced treatment system is used how is nutrient application on cropland impacted. Is more mineral fertilizer added to cropland to replace the manure nutrients being treated, is manure processed into a fertilizer product that is easier to manage and utilize as a crop fertilizer, and how do we account for these potential environmental impacts in comparing the baseline and alternative manure treatment systems.

Similarly, regulations for small and medium sized feedlot operations are often predicated on the impact these operations have on water quality or their potential to impact water quality. This creates regulations with our true goal, waters meeting the quality standards set for their dedicated use, in mind. However, this often leads to confusion among cattle producers over their requirements as it makes the standard that the runoff control system must achieve unclear, i.e., there isn't a specific storm size or design standard provided, but instead a necessity to not impair water quality. One way of clearing up this confusion is to specify a specific design standard, such as the 25-year, 24-hour storm, that these facilities must meet. However, if our goal truly is to have all water meet the standards specified for their designated use, this methodology may not be the most efficient use of funds. Perhaps a better methodology is to classify operations based on a series of factors, including distance from the water body, size of the operation, landscape between the operation and the stream, etc. to classify operations into different groups based on the risk they pose to water quality. This would ensure that systems that provide the desired level of environmental security are being constructed where they are needed while providing cattle producers in less sensitive areas with a way to continue to utilize low cost systems. Alternatively, watershed scale modeling could be performed to evaluate the influence of differing levels of runoff control would have on water quality with the watershed; however, this would be challenging to implement limiting its practicality.

Vegetative Treatment Systems

A vegetative treatment systems (VTSs) are a combination of treatment components, at least one of which utilizes vegetation, designed to manage open lot runoff (Moody et al., 2006). Vegetative treatment areas (VTAs) and vegetative infiltration basins (VIBs) are two possible vegetative components for VTSs; other options include wetlands, serpentine channels, and settling benches. As vegetative treatment systems have developed and the technology matured, different variations and system configurations have developed. In general, all systems start with some sort of solids settling system, i.e., settling basins, settling benches, or similar settling structure where the coarse solids are removed by sedimentation. Effluent is then metered onto either a vegetative infiltration basin and then to the vegetative treatment area or directly onto the vegetative treatment area where water and nutrients are utilized for forage production. A VIB is a flat area, surrounded by berms, planted to permanent vegetation (Moody et al., 2006). A flood effect is used to distribute the effluent over its surface. Drainage tiles located 1 to 1.2 m (3.4 to 4 ft) below the soil surface to encourage infiltration of effluent. The tile lines collect effluent that percolates through the soil profile; captured effluent is then pumped onto a VTA for further treatment.

Vegetate treatment area types included sloped, level, pumped, and sprinkler VTAs. A sloped VTA is an area level in one dimension, with a slight slope along the other dimension, to facilitate sheet flow that is planted and managed to maintain a dense stand of vegetation (Moody et al., 2006). Operation of a sloped VTA consists of applying solid settling basin effluent (or vegetative infiltration basin effluent) uniformly across the top of the vegetated area and allowing the effluent to sheet-flow down the slope (Moody et al., 2006). A pumped VTA is similar to that of a sloped VTA; however, since the effluent is pumped to the VTA there is greater flexibility in its location, i.e., it can be located at an elevation above the feedlot since effluent transport is mechanized. As with a sloped VTA, sheet flow is used to distribute the effluent over the length of the VTA. Similar location flexibility exists for a sprinkler VTA; however, in this case rather than relying on flow to distribute the effluent, a sprinkler system is utilized (Gross and Henry, 2007). This allows more precise control over effluent distribution, but a greater degree of effluent pretreatment is needed to limit sprinkler clogging and abrasion of the effluent distribution equipment. A level VTA is similar to sloped, but in this case the VTA is level in both directions and utilizes shallow ponding, i.e., a flooding effect, to distribute the effluent over the VTA. This is similar to a VIB, but no drainage tiles are present to encourage infiltration.

Vegetative Treatment Systems: Theory and Concept

The theory behind all runoff control systems developed thus far has been similar; break the hydraulic connection between the feedlot and the water of concern. This perspective can be seen in the work of Smith et al. (2007) who states "if no water is released, no nutrients will be released either." Although this is sound in principle, creating a "no-discharge" system has proven elusive, with researchers suggesting that under certain weather conditions discharges are likely to occur (Koelliker et al., 1975; Sensink and Miner, 1975; Zovne et al., 1977; Wulf and Lorimor, 2005; Andersen et a., 2010).

Moreover, Moffitt and Wilson (2004) point out that the waste management system is only as good as the operators' ability to follow their operating/nutrient management plans, which can be impacted by the field conditions on which the containment structures contents were to be applied. Vegetative treatment systems are no exception, even with proper siting and management certain weather conditions, i.e., large, intense storms or prolonged rainy periods, may cause a release.

In theory, vegetative treatment systems provide several advantages over their containment basin-land application systems. Most notably, as vegetative treatment systems seek to minimize long-term effluent storage, making them less prone to catastrophic failures, such as berm breaks. This provides a significant advantage, because although catastrophic failures of containments basins are relatively rare, when they do occur they generate a large amount of negative publicity due to the environmental harm they can cause. However, by minimizing long-term effluent storage, vegetative treatment systems become more sensitive to short-term weather patterns. Another key difference between vegetative treatment systems and containment based systems is that when effluent is released from the vegetative treatment system it has already received some treatment as it has flowed through a settling basin, a vegetative treatment area, and possibly an infiltration basin. Effluent released from a containment basin system has only received treatment from a settling basin. Providing a greater degree of treatment of released effluent provides a failsafe for the vegetative treatment system, provides increased environmental security should a release occur, and breaks the sedimentological connection between the feedlot and the surface water.

One limitation of vegetative treatment systems is that since they serve as the final effluent disposal area nutrient cycling (nitrogen and phosphorus) within the treatment system is much more critical to understanding the treatment mechanism and sustainability of the system than it is for a containment-land application system. Specifically, when we design containment-land application systems our focus is on understanding hydraulics, i.e., what size the basin must be to contain all runoff between pump-out periods. With vegetative treatment systems, it is still important to understand the hydraulics of the system, but since, in this case, the VTS also serves as the application area, designs must also consider nitrogen and phosphorus budgets. Hydraulic, phosphorus, and nitrogen budgets will be discussed conceptually in the following sections.

Hydraulic Considerations in VTS Design, Management, and Siting

Understanding the hydraulic budget of the vegetative treatment systems is essential to minimize unplanned releases, for evaluating leaching potentials, and to understand when aerobic/anaerobic (oxidizing/reducing) conditions occur in the soil profile. As was stated previously, if no water is released from the runoff control system then no nutrients will be either. Thus the question we seek to address here is how do we design, site, and manage vegetative treatment systems such that no release will occur from events smaller than the design storm. Several methodologies have been proposed; some revolve around detailed simulation modeling that evaluate how a series of different precipitation events cause differing hydraulic responses (Wulf and Lorimor, 2005; Andersen et al., 2010; Tolle, 2009), while others have suggested developing general rule-of-thumb sizing guidelines or sizing for a one-time occurrence of the design storm is appropriate (Blume, 2006). From a practical perspective all both methodologies are useful at certain times. Although detailed simulation modeling allows the most focused effort to maximize system performance while minimizing cost, it does so at the expense of great time and energy investments in setting up and running the simulation as well as in obtaining the necessary inputs to ensure model accuracy. Alternatively, general guidelines on siting and sizing VTSs provides estimates of system requirements, but may fail to produce acceptable systems under certain circumstances.

Given these risks we suggest that were feasible, and when robust estimates of system performance are required, simulation modeling be performed. However, it is our intention here to provide recommendations based on the general guideline approach, focusing on big-picture concepts. One thing our work has taught us is that given certain weather conditions, vegetative treatment system effluent releases will occur; when they do it is important to manage the system such that the release is a result of the rainfall onto the VTA rather than the runoff from the feedlot surface. VTS system performance is greatly enhanced when the operator has the ability to actively manage when the feedlot runoff is distributed to the VTA. Additionally, the use of properly designed physical flow barriers, such as berms, or the use of effluent recycling systems can limit or eliminate releases as a result of chronic wet periods. With these ideas in mind we think there are three major considerations for optimizing vegetative treatment system performance on a hydraulic basis; these are siting, design, and management. The impact of each of these three factors is discussed below.

SITING

Siting is of premier importance for achieving successful runoff control. Specifically, hydraulic conductivity and depth to water table play large roles in the system's hydraulic performance. Sites with shallow water table are often hydraulically challenged as there is less pore-space available in the soil profile to infiltrate and store additional water. This makes the sites more susceptible to releases via saturation excess flow, i.e., saturation of the soil profile from the bottom up. At locations with deeper water tables this phenomenon was less likely to occur. At these locations the primary mechanism of vegetative treatment area release is Hortonian flow, i.e., the application rate exceeded the infiltration rate of the soil. Of these two issues, management, i.e., controlling the effluent application rate can minimize or eliminate Hortonian flow from the applied feedlot runoff, where as runoff due to saturation excess is much more challenging to manage for (often requiring longer storage times before effluent application can commence). This makes selecting sites with appropriate depths to water table critical for achieving the desired performance.

Before suggesting required groundwater depths a few notes of caution. The groundwater depth guidelines provided here are only from a surface release perspective, not for preservation of groundwater quality. It has been our experience (five years of monitoring on six sites) that groundwater will not be negatively impacted from vegetative treatment systems, but long-term (>10 monitoring) should be conducted at numerous sites to verify this for sites under different hydrologic and geographic. With that said, minimum distance to groundwater can be estimated based on a specified design storm size and soil type following the principles laid out in Andersen et al. (2010). The design storm size specified here is 13 cm (5.1 inches), which is approximately the 25-year, 24-hour storm for much of Iowa. There must then be sufficient pore space in the soil profile to infiltrate this depth of precipitation, i.e., the current air filled porosity of the soil must be equal to the design storm. In performing this analysis, we assumed a hydrostatic soil moisture profile with the water table at a specified depth. In performing the calculation several soil properties were required, these included the porosity, the field capacity, the air entry pressure, and the pore size distribution index. These soil properties were estimated using the regression equation presented by Saxton and Rawls (2006) based on soil texture. For each soil texture representative sand and clay contents were selected and the required soil properties calculated. The required water table depth to have sufficient air-filled porosity to infiltrate 13 cm of precipitation was calculated. The results for various soil textures are shown in Table 1. As can be seen, these depths ranged from 1.5 m (4.9 feet) to 3.8 m (12.5 feet) depending on the soil texture.

Table 1. Required water table depth to have sufficient air-filled pore space to store 13 cm of water.

Soil Type	% Sand	% Clay	Required Water Table Depth m (ft)
Clay	30	50	3.8 (12.5)
Clay Loam	33	30	3.3 (10.7)
Loam	42	18	2.7 (9.0)
Loamy Sand	82	6	1.6 (5.3)
Sand	92	5	1.5 (4.9)
Sandy Clay	52	42	3.6 (11.8)
Sandy Clay Loam	60	28	2.6 (8.7)
Sandy Loam	65	10	2.0 (6.5)
Silt	7	6	3.7 (12.3)
Silty Clay	7	47	3.6 (11.7)
Silty Clay Loam	10	34	3.6 (11.8)
Silt Loam	20	20	3.4 (11.3)

Table 1 listed the required depth to groundwater to have available space to infiltrate direct rainfall onto the VTA; however, for a VTS to be successful it must also have space in the soil profile to store runoff from the feedlot. Again, assuming a 13-cm design storm the volume of runoff from the feedlot can be estimated using the SCS curve number method (~91). This would result in 10 cm of runoff from the feedlot. The effect of this runoff on the required water table depth can be minimized by storing the effluent into a containment basin until the water table level in the VTA has receded; however, if the producer wishes to release this effluent onto the VTA during or shortly after the storm, the required depth of the water table would increase. In this case the required depth is a function of two parts, the required depth to infiltrate all direct rainfall onto the VTA (which was presented in Table 1), and the depth required to infiltrate and store the feedlot runoff in the soil profile. This results in the required water table depth being a function of the VTA to feedlot area ratio as well as soil type. The results are shown in figure 1 for three example soils: clay, loam, and sand. As can be seen, the required depth increases rapidly at VTA: feedlot area ratios less than one, but at ratios above one the required depth is relatively stable.

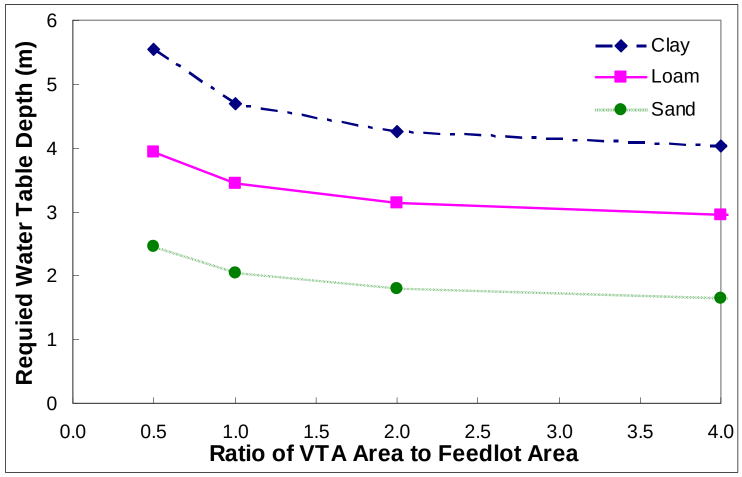

Figure 1. Water table depth requirement as a function of the ratio of VTA area to feedlot area.

In addition to the water table depth, the rate at which water can be utilized or leached by the vegetative treatment system also plays a crucial role in determining whether a VTA will operate successfully. This water utilization can either be by evapotranspiration or leaching, but it represents some sort of loss of water from soil zone. This rate is presumably a function of the evapotranspiration rate and the soils hydraulic properties (hydraulic conductivity and pressure gradients) and thus would be different for every soil type and location. However, in a general sense, we would expect a relatively similar water utilization pattern to emerge across regions with similar climates. To test this hypothesis, the percent hydraulic control (the percent of water added to the VTA through precipitation and feedlot runoff application that was not released from the VTA via surface outflow) was plotted against the hydraulic loading rate (amount of water added to the VTA per operational day). The results were remarkably consistent, a linear relationship between percent hydraulic control and the hydraulic loading rate. This pattern would seem to indicate that there is a critical loading rate, of about 0.25 cm/day (figure 2), which is approximately that average potential evapotranspiration rate throughout much of Iowa. We then proceeded to determine what VTA: feedlot area ratio would be required to achieve this loading rate at different locations throughout Iowa (based on average annual precipitation at the location). The hydraulic loading rate again stabilized a VTA: feedlot ratio of around 1:1 (figure 3). Based on this result we recommend VTAs be constructed to be at least the size of the feedlot.

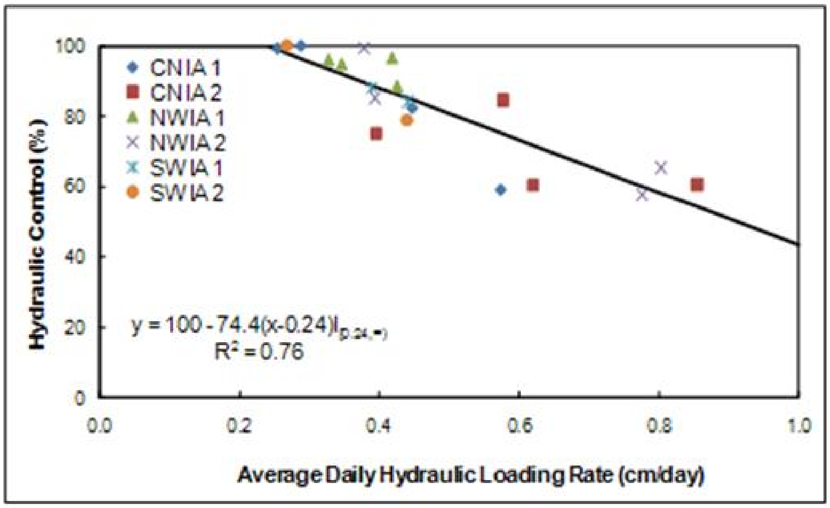

Figure 2. Plot of VTA hydraulic control versus the VTA's average daily hydraulic loading rate. Each point represents that average of one year of data.

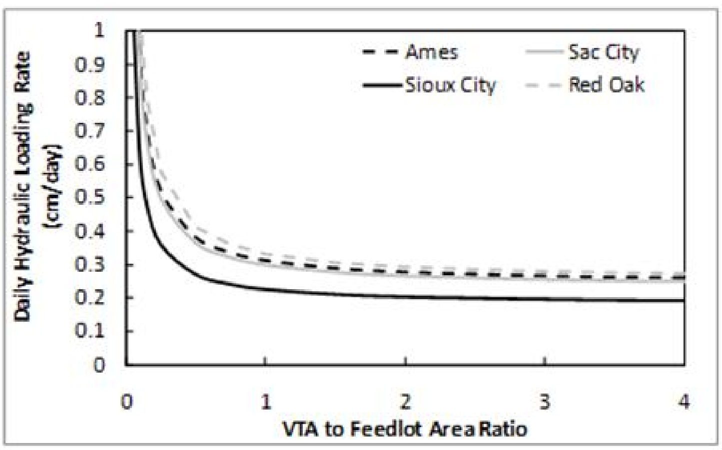

Figure 3. Predicted average daily hydraulic loading rate versus the vegetative treatment area to feedlot ratio. Feedlot runoff calculated using a curve number of 91 for the feedlot.

DESIGN

Designing VTS systems for ease of management and operational flexibility is a key factor in vegetative treatment system performance. Based on our monitoring results and experiences with these systems we recommend that all solids settling basin outlets have a control structure that allows the producer to close off the basin outlet and temporarily detain runoff effluent. This is critical at sites with shallow water tables where soil saturation can occur; in areas with deeper water tables this management practice has been effective in reducing the risk of a vegetative treatment system release. In addition to providing a delay before effluent is applied to the vegetative treatment area, a control structure also improved both the performance and consistency of solids removal during treatment in the settling basin.

The solid settling basin needs to be constructed to facilitate period solids removal. To facilitate cleaning, the design the basin should be designed to accommodate the available equipment. Additionally, the designer needs to consider the climatic conditions of the region. In Iowa, producers have found it challenging to get the basins cleaned out, often times there was only a short window available in the summer in which the basin was dry enough to drive equipment into. In drier climates this has seemed to be less of an issue. If the VTS is located in a region where clean-out could prove difficult it may be advantageous to line portions of the basin with concrete so that producer can clean out solids during wetter periods.

Designs which take into consideration even distribution of the feedlot runoff onto the vegetative treatment area perform better. Several methods have been suggested, including gated pipe, level-lipped spreaders, and sprinkler irrigation. Sprinklers have the advantage of providing uniform distribution over the entire vegetative treatment area, but have additional costs associated with their use (i.e., pumps, application equipment, piping). Gated pipes and level-lipped spreaders are often used on sloped vegetative treatment areas where gravity flow can distribute the effluent down the length of the vegetative treatment area. Gravity distribution has the disadvantage of accumulation of nutrients at the upper end of the VTA. This occurs because small storm events do not generate sufficient runoff and flow rates to move the feedlot runoff through the entire length of the treatment area. To some extent this can be alleviated by temporally detaining runoff within the settling basin and then surging the stored effluent onto the VTA, but even with this technique the application depth will still be deeper at the entrance to the treatment area. VTA's with gravity flow distribution require maintenance to ensure that uniform sheet flow occurs and to avoid flow channeling. Visual inspections can often be used to diagnose if channeling is occurring; if it is topsoil fill should be added to low area, the fill should be seeded, and the channel rested until vegetation is estabillished.

Perhaps the most important design variable is the vegetative treatment area size. There is no simple criteria that can be used to provide a single answer for ideal size as it is affected by climate, siting conditions, and the management of the system. However, based on our monitoring results and some simple water balance modeling we feel that a feedlot to vegetative treatment area ratio of one-to-one provides a good compromise between size and operational flexibility.

We recommend that a small berm or end block (less than 2 feet in height) be constructed at the end of the vegetative treatment area for overland flow designs. The observation of the authors is that this added measure minimized release of effluent that could not be infiltrated by the treatment system during a distribution event. This berm should have an emergency overflow and a means in which the producer could dewater the ponded area to preserve vegetation. Varying soil conditions such as antecedent moisture conditions in the VTA and vegetation retardance vary daily, and make it difficult to know when to stop a release from a basin to a VTA. There is a delayed reaction between stopping a basin release and when a wetting front ceases its movement down the VTA. This berm or end block provides insurance for the producer should they misjudge and apply a little too much effluent or apply it a little too quickly. This ponding provides the system operator with feedback on the system management, that is if there is a lot of water ponded at the end of the VTA, the system operator knows that too much effluent was released or release occurred for too long of period. Adjustments to management can then be made during the next application event. Eventually experience is gained and they (the system operator) are able to judge the correct volume to release from the basin. The purpose of this berm is not to catch all vegetative treatment area runoff and hold it for extended periods of time, but rather act as a safety measure to make system management easier. One critical design issue for this berm or end block is that during heavy rain storms rainwater runoff the VTA is collected. This runoff is substantially cleaner than runoff originating from the feedlot and future consideration should be given to whether this VTA only runoff needs to be recycled, infiltrated, or could be released from the systems.

MANAGEMENT

Based on our monitoring results it is clear that infiltration provided the majority of treatment in the vegetative treatment systems studied. Maximizing infiltration in VTA's are key to their success in minimizing releases of feedlot runoff effluent. This puts a premium importance on proper management of the system. Systems that provide the producer with control over when and at what rate effluent is applied to the vegetative treatment components offer the producer with the opportunity to maximize treatment. It has been our experience that learning the most effective management techniques can take several years, but certainly can have a positive impact on nutrient mass releases. Moreover, one management technique cannot be recommended for all systems; there will be a learning curve as producers experiment with their systems to see what works best for their operation, their management style, and the various weather conditions they encounter. However, there are several management recommendations that can be generalized to all sites.

- Producers must be vigilant in watching for signs of flow channelization and maintaining uniform sheet-flow over the vegetative treatment area. Gullies and rills must be repaired by filling and reseeding the areas.
- System components (level spreaders, settling basins, etc.) should be cleaned as often as weather permits.
- Good vegetation is critical to success. Vegetation stands can take several years to develop, but improve soil structure, increase infiltration, and provide increased resistance to flow as the stand improves.
- Settling basin effluent should be captured and held until after a storm event. Allowing for a day or two to pass until distribution to the VTA improves performance. This can be achieved with a valve on the settling basin outlet(s). All effluent should be released to the VTA within 72-96 hours to accommodate the next event. While not a storage system, the researchers found this delay a very useful tool in improving the performance of VTS systems.
- Provide mechanisms for the producer to adapt and manage the VTS. Valves/flow metering devices were found to be an important management tool for owners and operators. Timers and surge valves can be utilized on pumped systems. Provide producers with the ability to adapt to weather conditions and adjust application rate to the current soil conditions.
- Allow for distribution of sediment basin effluent to cropland. If effluent is land applied to crop land during typical manure application periods (spring and fall) it reduces hydraulic loading to the vegetative treatment system during critical periods when evapotranspiration rates are lower.

Phosphorus Cycling and Retention in VTSs

Under normal agricultural management practices phosphorus is transferred to surface waters due to mobilization and delivery from a phosphorus source (Haygarth et al., 2005). In our case the phosphorus source is the feedlot surface; during precipitation events particulate phosphorus is eroded from the feedlot surface and transported to the runoff control system. Additional phosphorus on the feedlot surface is solubilized during the runoff event and transported in dissolved form. The runoff water from the feedlot is detained in a settling basin where much of the particulate phosphorus is captured; however, little to no dissolved phosphorus is retained. The phosphorus containing wastewater is then applied to the vegetative treatment area, which treats the wastewater by modifying phosphorus delivery to surface waters by retaining the applied phosphorus via physical retention of any remaining particulate phosphorus and by geochemical and biological retention (sorption, precipitation, assimilation) of dissolved phosphorus. However, when phosphorus application exceeds removal with crop residues, it accumulates over time, resulting in phosphorus enrichment in the soil, and potentially remobilization of the retained phosphorus. Understanding how vegetative treatment systems modify phosphorus delivery from the feedlot, what controls phosphorus retention within the treatment area, and when vegetative treatment areas become a source of dissolved phosphorus is critical for understanding their sustainability and life expectancy.

Physical retention of phosphorus has been one of the most studied mechanisms of phosphorus retention. It occurs as particulate phosphorus in the feedlot runoff settles or is filtered out of suspension by the dense vegetation in the treatment area. The vegetation increases surface roughness, slowing flow and causing sediment deposition. The fibrous root system of the perennial vegetation encourages infiltration by increasing permeability and porosity of the soil. This encourages increased infiltration and contact between the dissolved phosphorus and the soil particles, allowing time for geochemical retention, including sorption, precipitation, and biological uptake. Sorption/desorption processes are governed by the concentration of phosphorus in solution. The point where sorption and desorption are equal is called the equilibrium phosphate concentration (EPC_0) and is considered a key indicator of phosphorus leaching potential as it provides an indication of phosphorus solubility in the soil.

Thus far research has shown that when designed on a hydraulic basis phosphorus application greatly exceeds phosphorus removed with harvested vegetation (often by 5-10 times). This results in rapid accumulation in the surface soil; however, after five years of monitoring only limited vertical transport of phosphorus has been detected in the vegetative treatment areas. One technique that has been used to evaluate the status of phosphorus in soils and the soil’s future ability to sorb additional phosphorus is lab scale phosphorus isotherm techniques (Hu et al., 2006). In this technique soil is equilibrated with phosphorus solutions of differing concentration to evaluate how the phosphorus partitions between the soil and liquid phases. It has been proposed that this technique can be used to evaluate how much phosphorus can be added before the soil becomes saturated, i.e., before it will no longer sorb and retain additional phosphorus inputs. For example this technique was used by Hu et al.

(2006) to estimate how many years a land application area would provide phosphorus retention at a municipal wastewater treatment plant and by Baker et al. (2010) to estimate saturation lives of several VTAs. However, further evaluation by Andersen et al. (2011, Chapter 7 here) showed this technique wasn't viable as VTA soils that had received effluent application for five years and accumulated sufficient phosphorus to be near their saturation point exhibited increased capacity to sorb additional phosphorus as compared to the native soil. Roberts et al. (2011) reported that a similar phenomenon i.e., increased phosphorus sorption capacity, has occurred in many vegetated buffer strips at field edge boundaries. Despite these increases in phosphorus sorption capacity the soil's equilibrium phosphorus concentration have also increased significantly (Andersen et al., 2011, Chapter 7 here). Although not a perfect approach, it appears that the phosphorus sorption – phosphorus loading estimate generated using Baker's approach should provide a conservative estimate of phosphorus life.

Based on these results we suggest that future research on phosphorus cycling in vegetative treatment systems focus on evaluating how water-extractable phosphorus and the soil's equilibrium phosphorus concentration are impacted by continued use as an effluent disposal area. Being able to predict when these parameters reach critical thresholds holds the key to evaluating the life expectancy of the treatment system; however, at this time there doesn't appear to be a reliable method to make this prediction. With that said, it is clearly important to design vegetative treatment systems with some semblance of a phosphorus balance in mind. Although it may not be necessary to balance phosphorus application with crop removal, designers should strive to minimize the difference between these parameters to slow the rate of phosphorus accumulation in the soil profile. Based on our experience we'd suggest that vegetative treatment areas should be at least the size of the feedlot and as a first approximation we recommend using the phosphorus life approach outlined in Baker et al. (2010). Additional efforts to ensure good settling and to improve phosphorus removal in the settling basin are necessary to ensure long-term effectiveness of the system.
Based on our experience we provide the following suggestions for both operating and managing successful vegetative treatment systems in regards to phosphorus management.

- Settling basin effluent should be captured and held until after a storm event. Allowing a day or two to pass until distribution to the VTA improves performance and reduces phosphorus loading to the vegetative treatment area by allowing more time for sediment deposition. Other pretreatments that have the potential to remove phosphorus prior to application should be considered; these could include additions of polymers to increase solids capture, the use of alternative solids separation techniques, or even chemical treatment of the effluent to generate struvite.

- Good vegetation is critical to success; this vegetation not only slows the flow and improves soil structure and infiltration, but its harvest provides the only acceptable method of phosphorus removal. Reed canarygrass appears to have greater potential for phosphorus uptake than other grasses; where possible species with high phosphorus uptake rates should be utilized.
- VTA designs should consider using multiple channels and allow the producer to determine which channels are receiving effluent. This would allow the producer to continue utilize the treatment system while being able to dry and harvest vegetation from one of the channels, encouraging phosphorus removal.
- Producers must be vigilant in watching for signs of flow channelization and maintaining uniform sheet-flow over the vegetative treatment area. Gullies and rills must be repaired by filling and reseeding the areas. This will improve hydraulic and phosphorus distribution over the VTA area limiting the formation of hot spots.
- Soils provide the majority of phosphorus retention in the system. Selecting sites with an ability to sorb and fix large amounts of phosphorus is key to extending the life of the system.
- Methods that improve effluent distribution down the length of the VTA should be considered. Options include both sprinkler systems and surging effluent onto the VTA to distribute effluent more evenly over the length of the treatment area.

Nitrogen Cycling and Retention in VTSs

At present only a few studies have reported nitrogen balances from vegetative treatment systems. Woodbury et al. (2003) reported that at their system in Nebraska nitrogen removal with vegetation harvest exceeded that applied in the feedlot runoff; however, at six vegetative treatment systems at commercial operations in Iowa nitrogen application was in excess of crop removal (see table 2). We anticipate that most vegetative treatment systems on commercial operations will have nitrogen budgets more similar to those monitored on the Iowa systems than the Woodbury et al. (2003) system, as producers have tended to have their system sized on a hydraulic, rather than nutrient, budget basis. As nitrogen application is expected to exceed crop utilization its ultimate fate is of upmost importance in understanding the environmental impacts the runoff control system has. In order to better understand the fate of the applied nitrogen Andersen et al. (Chapters 8 and 9) performed several additional analyses. These included monitoring of groundwater quality beneath the treatment system to evaluate if nitrogen leaching, as either nitrate or ammonium, was occurring and significantly impacting shallow groundwater and a biological fractionation of soil organic matter to assess if nitrogen is accumulating in the soil profile and if organic matter is becoming nitrogen enriched.

These studies provided some indication of what may be happening within the treatment system and how their continued use as vegetative treatment systems is impacting nutrient cycling. What we observed was that while carbon-to-nitrogen ratios are decreasing in the soil organic matter, often a sign that nitrogen saturation is occurring, nitrate concentrations in groundwater decreased below background concentration, i.e., before system operation and use levels. However, decreased carbon-to-nitrogen ratios in the organic matter can also indicate that a greater amount of nitrogen may be lost as nitrous oxide if denitrification is occurring.

To better understand the results, a rough nitrogen mass balance is presented here. Measurements of nitrogen inputs applied to the vegetative treatment area and surface flow nitrogen outputs from the vegetative treatment area were determined by flow monitoring and concentration sampling from each effluent application or release. Similarly, nitrogen removal with vegetation harvest was determined based on harvested mass and vegetation nutrient concentration sampling as described in Andersen et al. (2011). The amount of nitrogen leached was determined based on a hydraulic balance and monthly sampling of groundwater in the vegetative treatment area. Inorganic nitrogen accumulation was estimated for the top 1.2 cm of the soil by taking the difference between background (2006) and 2009 soil samples from the vegetative treatment areas and multiplying by soil bulk density to convert to nitrogen accumulation to a unit area basis. Finally, organic nitrogen accumulation was estimated based on changes in mineralizable nitrogen; however, this estimate only accounts for the top six inches of the soil profile. Finally gaseous emissions were estimated as the difference in nitrogen inputs and outputs from the described nitrogen pools. The results are shown in table 2.

This methodology resulted in estimated nitrogen emission ranging from 500-1100 kg N/ha-yr and between 30-120% of the nitrogen inputs (one site exhibited substantial decreases in mineral nitrogen within the soil profile). These emissions could be the result of ammonia volatilization, nitrous and nitric oxide emission during nitrification, and nitrous oxide or diatomic nitrogen emission during denitrification, with presumably ammonia volatilization and denitrification emissions accounting for the majority of the gaseous nitrogen losses. Thus far no researchers have monitored nitrogen emission from the vegetative treatment area. Given the estimated magnitude of the emission predicted by our nitrogen mass balance it is imperative that these emissions be assessed to determine the amounts and forms of nitrogen emitted and management practices to minimize these emissions.

To evaluate the possibility of emission of this magnitude, nitrogen emissions from animal waste lagoons were surveyed. Results reported in literate showed considerable variation ranging from 219 (Aneja et al., 2001) – 57,670 (Zahn et al., 2001) kg NH_3-N per ha per year. Similarly, Liang et al. (2002) suggested, based on a modeling approach, that ammonia nitrogen emissions for a typical swine manure lagoon in North Carolina would be 2340 kg N/ha-yr. This is 2-4x the gaseous emission suggested in our balance and provides some indication that emissions of this magnitude are possible, especially considering that the effluent is surface applied. A recent study by Johns et al. (2011) on the nitrogen balance at a tomato canary wastewater land application area indicated gaseous nitrogen losses of 1500 to 2600 kg N/ha with often significant losses of native soil N occurring. These losses are similar to those suggested by the nitrogen balance on the vegetative treatment systems reported here, again indicating that nitrogen emissions of this magnitude are plausible. Other researchers (Hooda et al., 2003; Russel et al., 1993; and Fedler and Green, 2006) have reported denitrification rates of 200-1700 g N/ha-d, 12-240 g N/ha-d, and 524-2229 g N/ha-d respectively for municipal treated wastewater on a clay soil, meat processing wastes in a forest soils, and municipal treated wastewater at a land application area in Lubbock, Texas. These results range from 4 – 621 kg N/ha-yr and indicate that it is plausible that all to half of the estimated nitrogen emissions could be accounted for by denitrification with the remainder presumably emitted as ammonia. However, based on the narrowing C:N ratios in our labile soil organic matter it is probably that much of the nitrogen emitted during the denitrification process would be as nitrous oxide, and not diatomic nitrogen gas.

Unfortunately, given the current state of knowledge on nitrogen emissions from vegetative treatment systems it is not possible to provide siting, design, and management techniques at this time to minimize nitrogen emissions as ammonia and nitrous oxide while maximizing conversion to nitrogen gas. Given the magnitude of nitrogen being emitted from these systems, based on the nitrogen balance approach, it is imperative that this research be conducted to ensure that they are providing the desired environmental protection to air as well as water resources.

Conclusions

Vegetative treatment systems have proven to be a useful tool for feedlots looking to improve their environmental stewardship. Research at commercial feedlot operations over the last ten years has shown that proper siting, sound design, and active management or critical to the success of these systems. It is critical that minimum distances to groundwater (~ 1.5 – 3.0 m) are maintained to ensure system releases do not occur except from events larger than the design storm. Vegetative treatment system designs should seek to provide flexibility so that the operator can adapt to current weather conditions, and the producer must be vigilant in operating the system to ensure channeling doesn't develop and that effluent is being applied at a rate the system is capable of handling. Future research focusing on managing these systems to optimize denitrification and quantify nitrogen emissions is required to ensure VTSs are providing adequate protection of both air and water resources.

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Table 2. Projected nitrogen mass balances at each of the six monitored vegetative treatment areas. The amount of nitrogen applied and surface released from the vegetative treatment area was measured by monitoring in and outflows and effluent concentration sampling. The amount of nitrogen removed in harvested vegetation was monitored using the methodology described in the manuscript on phosphorus retention. The amount of nitrogen leached was described in the groundwater manuscript. Soil organic N accumulation only accounts of nitrogen accumulation in the top 15.4 cm of the soil profile and was based on change in mineralizable nitrogen content. Accumulation of inorganic N accounts for change in NH_4-N and NO_3-N concentrations in the top 1.2 m of the soil profile. Estimated gaseous nitrogen emissions were calculated as the nitrogen not accounted for in the previous pools. The percent of applied nitrogen ending in each pool is shown in parenthesis.

Site	N Applied (kg N/ha-yr)	N Released in Surface Flow (kg/ha-yr)	N Harvested in Vegetation (kg N/ha-yr)	Estimated N Leached (kg N/ha-yr)	Soil Inorganic N Accumulation (kg N/ha-yr)	Soil Organic N Accumulation (kg N/ha-yr)	Estimated Gaseous N Emissions (kg N/ha-yr)
CN IA 1	1244	276 (22%)	221 (18%)	39 (3%)	76 (6%)	123 (10%)	509 (41%)
CN IA 2	778	176 (23%)	55 (7%)	2 (<1%)	0 (0%)	1 (0%)	544 (70%)
NW IA 1	2008	566 (28%)	265 (13%)	22 (1%)	-7 (0%)	245 (12%)	917 (46%)
NW IA 2	3577	2084 (58%)	144 (4%)	15 (<1%)	102 (3%)	152 (4%)	1080 (30%)
SW IA 1	891	189 (21%)	128 (14%)	14 (2%)	-558 (-63%)	51 (6%)	1067 (120%)
SW IA 2	994	83 (8%)	244 (25%)	11 (1%)	-19 (-2%)	36 (4%)	639 (64%)